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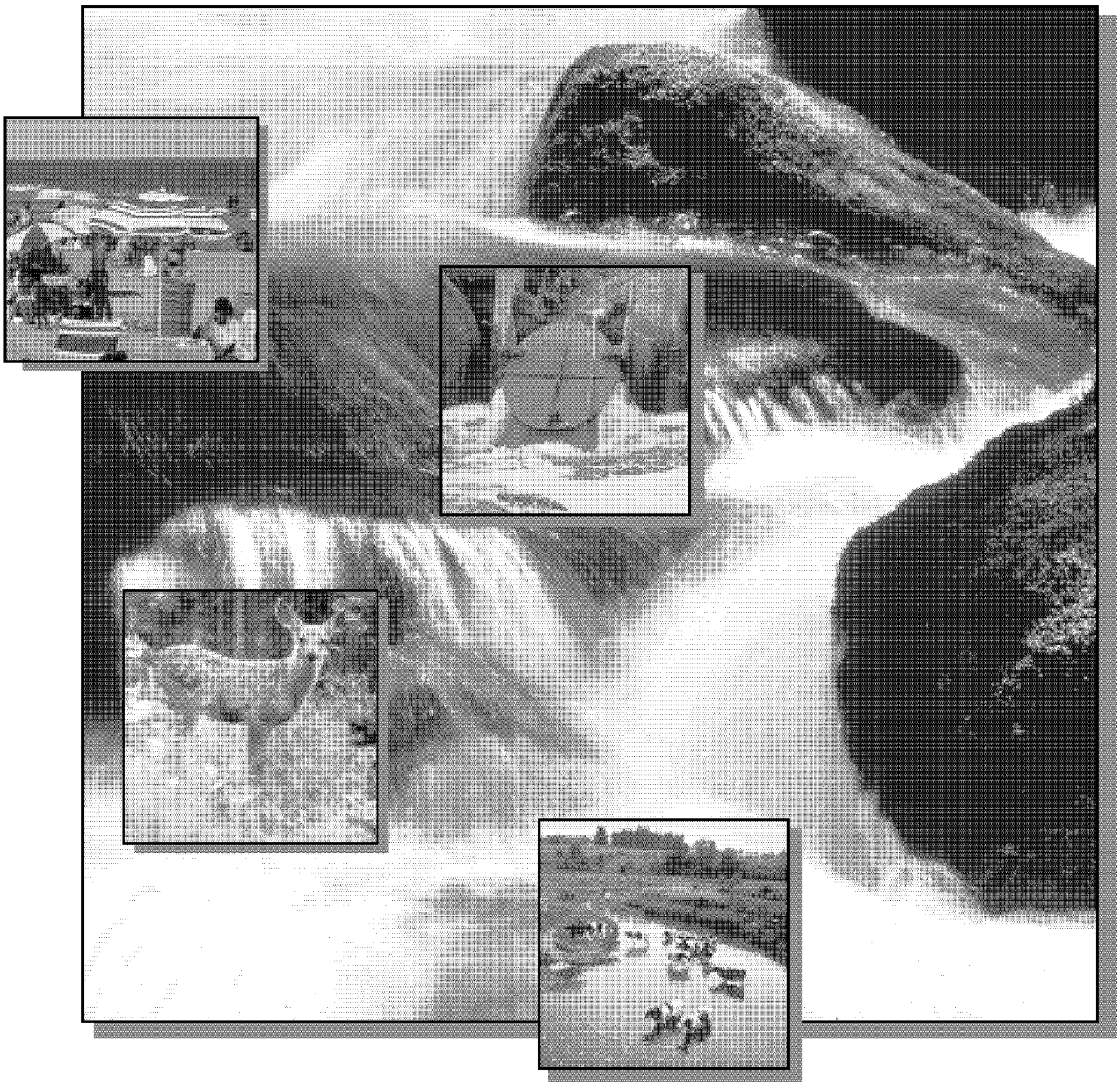
Protocol for Developing Pathogen TMDLs

First Edition



Acknowledgments

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Watershed Branch
Assessment and Watershed Protection Division
Office of Wetlands, Oceans, and Watersheds
Office of Water
United States Environmental Protection Agency
Washington, DC 20460

Foreword

Although many pollution sources have implemented the required levels of pollution control technology, there are still waters in the Nation that do not meet the Clean Water Act goal of "fishable, swimmable." Section 303(d) of the act addresses these waters that are not "fishable, swimmable" by requiring states, territories, and authorized tribes to identify and list impaired waters every two years and to develop Total Maximum Daily Loads (TMDLs) for pollutants in these waters, with oversight from the U.S. Environmental Protection Agency. TMDLs establish the allowable pollutant loadings, thereby providing the basis for states to establish water quality-based controls.

Historically, wasteload allocations have been developed for particular point sources discharging to a particular waterbody to set effluent limitations in the point source's National Pollutant Discharge Elimination System (NPDES) discharge permit. This approach has produced significant improvements in water quality by establishing point source controls for many chemical pollutants. But water quality impairments continue to exist in the Nation's waters. Some point sources need more controls, and many nonpoint source impacts (from agriculture, forestry, development activities, urban runoff, and so forth) cause or contribute to impairments in water quality. To address the combined, cumulative impacts of both point and nonpoint sources, EPA has adopted a watershed approach, of which TMDLs are a part. This approach provides a means to integrate governmental programs and improve decision making by both government and private parties. It enables a broad view of water resources that reflects the interrelationship of surface water, groundwater, chemical pollutants and nonchemical stressors, water quantity, and land management.

The *Protocol for Developing Pathogen TMDLs* is a TMDL technical guidance document prepared to help state, interstate, territorial, tribal, local, and federal agency staff involved in TMDL development, as well as watershed stakeholders and private consultants. Comments and suggestions from readers are encouraged and will be used to help improve the available guidance as EPA continues to build experience and understanding of TMDLs and watershed management.

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Preface

EPA has developed several protocols as programmatic and technical support guidance documents for those involved in TMDL development. These guidance documents, developed by an interdisciplinary team, provide an overall framework for completing the technical and programmatic steps in the TMDL development process. The *Protocol for Developing Pathogen TMDLs* is one of the three TMDL technical guidance documents prepared to date. The process presented here will assist with the development of rational, science-based assessments and decisions and ideally will lead to the assemblage of an understandable and justifiable pathogen TMDL. It is important to note that this guidance document presents a suggested approach, but not the only approach to TMDL development.

This document provides guidance to states, territories, and authorized tribes exercising responsibility under section 303(d) of the Clean Water Act for the development of pathogen TMDLs. This protocol is designed as programmatic and technical support guidance to those involved in TMDL development. The protocol does not, however, substitute for section 303(d) of the Clean Water Act or EPA's regulations; nor is it a regulation itself. Thus, it cannot impose legally binding requirements on EPA, states, territories, authorized tribes, or the regulated community and may not apply to a particular situation based upon the circumstances. EPA and state, territorial, and tribal decision makers retain the discretion to adopt approaches on a case-by-case basis that differ from this protocol where appropriate. EPA may change this protocol in the future.

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Introduction and Purpose of This Protocol

Objective: This Total Maximum Daily Load (TMDL) protocol was developed at the request of EPA regions, states[1], and tribes and is intended to provide users with an organizational framework for the TMDL development process for fecal pathogens. The process presented here will assist with the development of rational, science-based assessments and decisions and ideally will lead to the assembly of an understandable and justifiable TMDL.

Audience: The protocols are designed as tools for state TMDL staff, EPA regional TMDL staff, tribal TMDL staff, watershed stakeholders, and other agencies and private consultants involved in TMDL development.

OVERVIEW

Section 303(d) of the Clean Water Act provides that states, territories, and authorized tribes are to list waters for which technology-based limits alone do not ensure attainment of water quality standards (WQS). Beginning in 1992, states, territories, and authorized tribes were to submit their lists to EPA every two years. Beginning in 1994, lists were due to EPA on April 1 of each even-numbered year. States, territories, and authorized tribes are to set priority rankings for the listed waters, taking into account the severity of the pollution and the intended uses of the waters.

EPA's regulations for implementing section 303(d) are codified in the Water Quality Planning and Management Regulations at 40 CFR Part 130, specifically at sections 130.2, 130.7, and 130.10. The regulations define terms used in section 303(d) and otherwise interpret and expand upon the statutory requirements. The purpose of the *Protocol for Developing Pathogen TMDLs* is to provide more detailed guidance on the TMDL development process for waterbodies impaired because of pathogens.

EPA's regional offices are responsible for approving or disapproving state, territorial, or tribal section 303(d) lists and TMDLs, and for establishing lists and TMDLs in cases of disapproval. Public participation is to be provided for by states and tribes (or EPA regional offices, in the case of disapproval) when they establish lists or TMDLs.

In accordance with the priority ranking, states, territories, and authorized tribes are to establish TMDLs that will meet water quality standards for each listed water, considering seasonal variations and a margin of safety that accounts for uncertainty. States, territories, and authorized tribes are to submit their lists and TMDLs to EPA for approval and, once EPA approves them, are to incorporate these items into their continuing planning process. If EPA disapproves a state, territorial, or tribal list and/or TMDL, EPA will (within 30 days of disapproval and allowing for public comment) establish the list and/or TMDL. The state, territory, or tribe is then to incorporate EPA's action into its continuing planning process.

A TMDL is a tool for implementing state water quality standards. It is based on the relationship between sources of pollutants and in-stream water quality conditions. The TMDL establishes the allowable loadings for specific pollutants that a waterbody can receive without exceeding water quality standards, thereby providing the basis for states to establish water quality-based pollution controls.

A TMDL is the sum of the individual wasteload allocations for point sources and load allocations for nonpoint sources and natural background (40 CFR 130.2) with a margin of safety (CWA section 303(d)(1)(c)). The TMDL can be generically described by the following equation:

$$TMDL = LC = \sum WLA + \sum LA + MOS$$

where: LC = loading capacity,[a] or the greatest loading a waterbody can receive without exceeding water quality standards;
WLA = wasteload allocation, or the portion of the TMDL allocated to existing or future point sources;
LA = load allocation, or the portion of the TMDL allocated to existing or future nonpoint sources and natural background; and
MOS = margin of safety, or an accounting of uncertainty about the relationship between pollutant loads and receiving water quality. The margin of safety can be provided implicitly through analytical assumptions or explicitly by reserving a portion of loading capacity.

[a]TMDLs can be expressed in terms of mass per time, toxicity, or other appropriate measures.

[1]Note: The term "states" will be used to denote states, territories, and authorized tribes.

Guidance on developing TMDLs is readily available for many chemical pollutants. For some pollutants, however, the development of TMDLs is complicated because of the lack of adequate or proven tools or information on the fate, transport, or impact of each pollutant within the natural system. EPA is developing TMDL protocols to provide guidance on TMDL development. The protocols represent a suggested approach, but not the only approach to TMDL development. EPA will continue to review all TMDLs submitted by states pursuant to Section 303(d) of the Clean Water Act and 40 CFR 130.7.

The TMDL protocols focus on Step 3 (Development of TMDLs) of the water quality-based approach, depicted in Figure 1-1 (USEPA, 1991a; 1999). This specific step is divided into seven components common to all TMDLs, and each component is designed to yield a product that is part of a TMDL submittal document.

COMPONENTS OF TMDL DEVELOPMENT

The following components of TMDL development may be completed concurrently or iteratively depending on the site-specific situation (Figure 1-2):

- Problem identification
- Identification of water quality indicators and targets
- Source assessment
- Linkage between water quality targets and sources
- Allocations
- Follow-up monitoring and evaluation
- Assembling the TMDL

Note that these components are not necessarily sequential steps, but are provided more as a guide and framework for TMDL development. Although some of the submittal components (e.g., TMDL calculation and allocations) are part of the legally required TMDL submittal and others are part of the administrative record supporting the TMDL, this protocol considers each component equally.

Problem Identification

The objective of problem identification is to identify the key factors and background information for a listed waterbody that describe the nature of the impairment and the context for the TMDL. Problem identification is a guiding factor in development of the remaining elements of the TMDL process.

Identification of Water Quality Indicators and Target Values

The purpose of this component is to identify numeric or measurable indicators and target values that can be used to evaluate attainment of water quality standards in the listed waterbody. Often the TMDL target will be the numeric water quality criteria for the pollutant of concern. In some cases, however, TMDLs must be developed for parameters that do not have numeric water quality standards. When numeric water quality criteria do not exist, impairment is determined by narrative water quality standards or identifiable impairment of designated uses (e.g., degraded fishery). The narrative standard is then interpreted to develop a quantifiable target value to measure attainment or maintenance of the water quality standards.

Source Assessment

During source assessment, the sources of loading for the pollutant of concern to the waterbody are identified and characterized by type, magnitude, and location.

Linkage Between Water Quality Targets and Sources

To develop a TMDL, a linkage between the selected indicator(s) and target(s) and the identified sources must be defined. This linkage establishes the cause-and-effect relationship between the pollutant sources and the in-stream pollutant response and allows for an estimation of the loading capacity. Once defined, the linkage yields the estimate of total loading capacity, which is the maximum amount of pollutant loading (e.g., fecal indicators) a waterbody can assimilate and still attain without exceeding water quality standards. The relationship can vary seasonally, particularly for nonpoint sources, with factors such as precipitation.

Allocations

Based on the established target/source linkage, pollutant loadings that will not exceed the loading capacity and will lead to attainment of the water quality standard can be determined. These loadings are distributed or

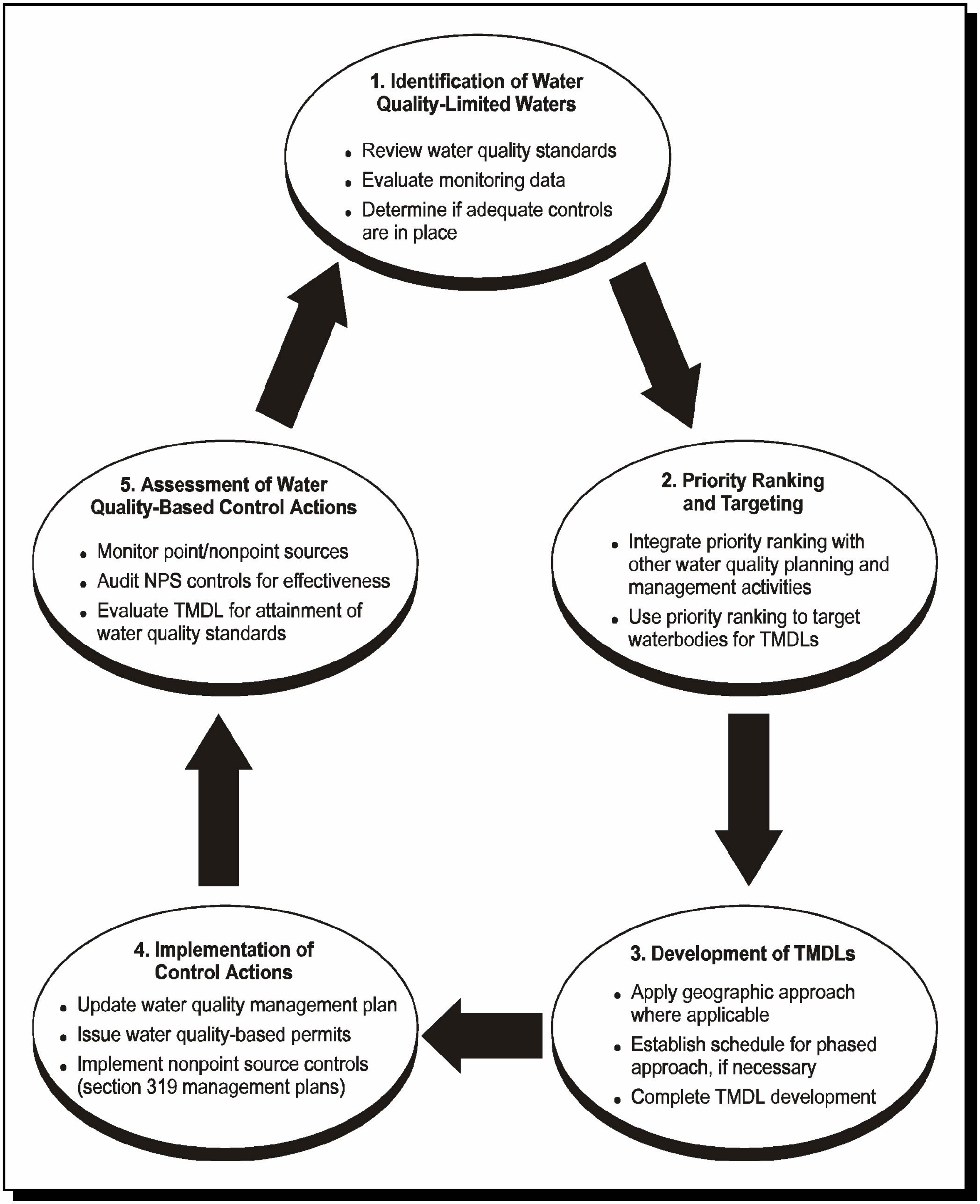

Figure 1-1. General elements of the water quality-based approach (adapted from USEPA, 1991a)

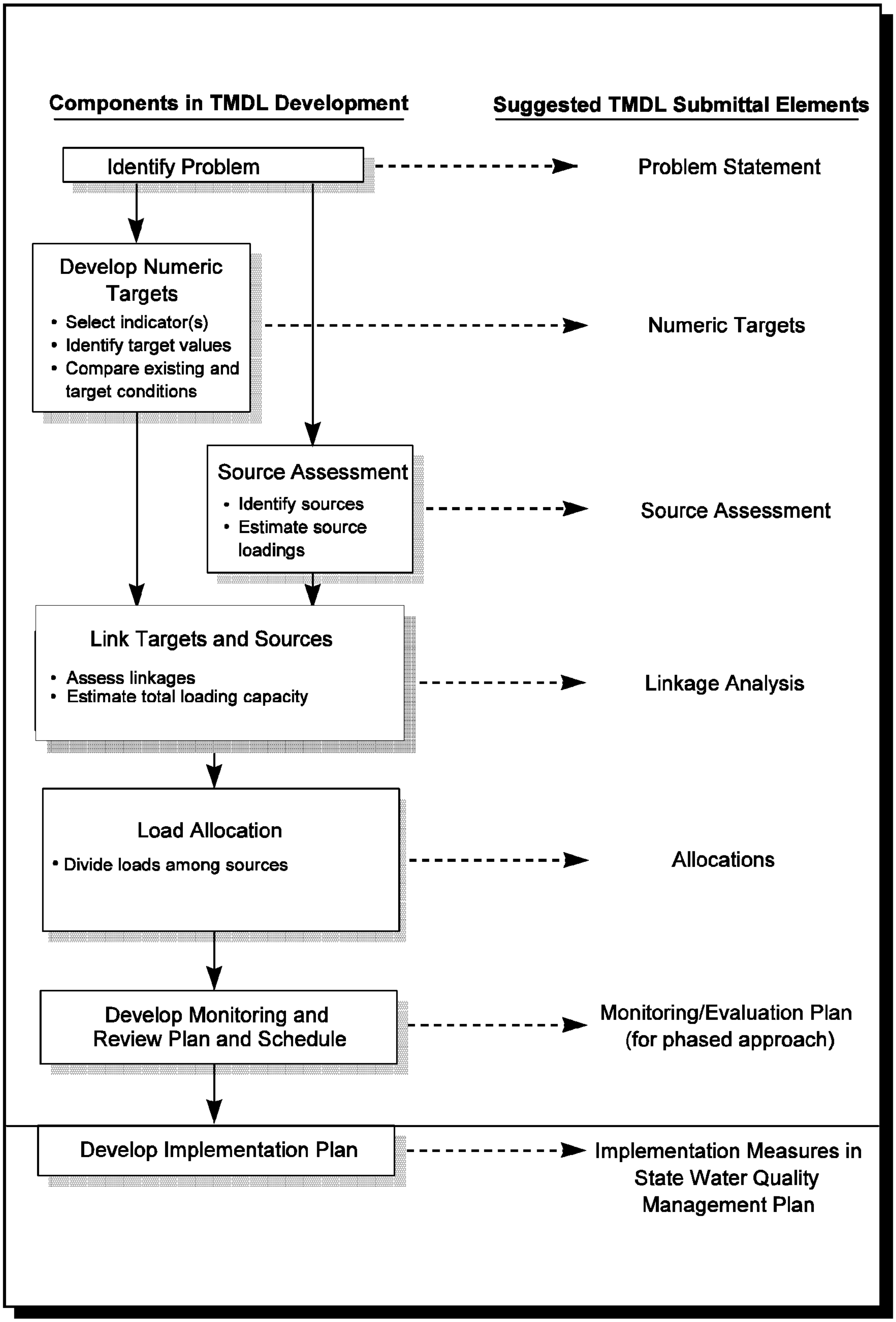

Figure 1-2. General components of TMDL development

"allocated" among the significant sources of the pollutant of concern. The allocations are a component of the legally approved TMDL. Wasteload allocations contain the allowable loadings from existing or future point sources, while load allocations establish the allowable loadings from natural background and from existing and future nonpoint sources. The margin of safety is usually identified during this step to account for uncertainty in the analysis, although it may also be identified in other TMDL components. The margin of safety may be applied implicitly by using conservative assumptions in the TMDL development process or explicitly by setting aside a portion of the allowable loading.

Follow-up Monitoring and Evaluation

TMDL submittals should include a monitoring plan to determine whether the TMDL has resulted in attaining water quality standards and to support any revisions to the TMDL that might be required. Follow-up monitoring is recommended for all TMDLs, given the uncertainties inherent in TMDL development (USEPA, 1991a; 1997a; 1999). The rigor of the monitoring plan should be based on the confidence in the TMDL analysis. A more rigorous monitoring plan should be included for TMDLs with greater uncertainty and where the environmental and economic consequences of the decisions are greatest.

Assembling the TMDL

In this component, those elements of a TMDL submittal required by statute or regulation are clearly identified and compiled, and supplemental information is provided to facilitate TMDL review.

For each component addressed in this protocol, the following format is used:

- Guidance on key questions or factors to consider.
- Brief discussions of analytical methods.
- Discussions of products needed to express the results of the analysis.
- Examples of approaches used in actual settings to complete the step.
- References on methods and additional guidance.

By addressing each of the seven TMDL components, TMDL developers can complete the technical aspects of TMDL development. Although public participation requirements are largely outside the scope of this document, early involvement of stakeholders affected by the TMDL is strongly encouraged because of the complex and often controversial nature of TMDLs. The protocols also do not discuss issues associated with TMDL implementation (note bottom of Figure 1-2). Methods of implementation, such as National Pollutant Discharge Elimination System (NPDES) permits, state nonpoint source (NPS) management programs, the Coastal Zone Act Reauthorization Amendments (CZARA), and public participation are discussed in *Guidance for Water Quality-based Decisions: The TMDL Process* (USEPA, 1991a, 1999) and in the August 8, 1997, memorandum "New Policies for Establishing and Implementing Total Maximum Daily Loads (TMDLs)" (USEPA, 1997a).

PURPOSE

This protocol provides a description of the TMDL development process for pathogens and includes case study examples to illustrate the major points in the process. It emphasizes the use of rational, science-based methods and tools for each step of TMDL development to assist readers in applying a TMDL development process that addresses all regulatory requirements.

Note that this protocol focuses mainly on fecal coliform bacteria as pathogen indicators since that is the indicator currently in most state water quality standards. However, EPA strongly encourages states that have not already done so, to adopt the recommendations set forth in *Ambient Water Quality Criteria for Bacteria* - 1986 or other water quality criteria for bacteria based on scientifically defensible methods into their water quality standards to replace water quality criteria for total or fecal coliforms. EPA's 1986 water quality criteria for bacteria recommend the use of enterococci for marine waters and *E. coli* or enterococci for fresh waters. It is also important to realize that the presence of indicator bacteria does not always prove or disprove the presence of human pathogenic bacteria, viruses, or protozoans.

References and recommended reading lists are provided for readers interested in obtaining more detailed background information. This protocol has been written with the assumption that users have a general background in the technical aspects of water quality management and are familiar with the statutory and regulatory basis for the TMDL program. A glossary is included at the end of the document with definitions of some commonly used terms.

RECOMMENDED READING

(Note that a full list of references is included at the end of this document.)

USEPA. 1991a. *Guidance for Water Quality-based Decisions: The TMDL Process.* EPA 440/4-91-001. U.S. Environmental Protection Agency, Washington, DC. <http://www.epa.gov/owow/tmdl/policy.html>.

USEPA. 1995a. *Watershed Protection: A Project Focus.* EPA 841-R-95-003. U.S. Environmental Protection Agency, Office of Water, Washington, DC.

USEPA. 1995b. *Watershed Protection: A Statewide Approach.* EPA 841-R-95-001. U.S. Environmental Protection Agency, Office of Water, Washington, DC.

USEPA. 1997a. New Policies for Establishing and Implementing Total Maximum Daily Loads (TMDLs). U.S. Environmental Protection Agency, Washington, DC. <http://www.epa.gov/owow/tmdl/policy.html>.

USEPA. 1999. *Draft Guidance for Water Quality-based Decisions: The TMDL Process.* 2nd ed. EPA 841-D-99-001. U.S. Environmental Protection Agency, Washington, DC. <http://www.epa.gov/owow/tmdl/proprule.html>.

General Principles of Pathogen Water Quality Analysis

Objective: To develop a pathogen TMDL, it is important to have a basic understanding of pathogen processes in a watershed and how excessive pathogens can affect water quality and designated uses of water. This section provides background information on pathogen impacts on designated uses, types of pathogens, pathogen sources and transport, indicator organisms, survival factors, and potential control strategies.

IMPACTS OF PATHOGENS ON DESIGNATED USES

Microorganisms are ever present in terrestrial and aquatic ecosystems. Most types are beneficial, functioning as agents for organic and synthetic chemical decomposition, as food sources for larger animals, and as essential components of the nitrogen cycle and other biogeochemical cycles. Some reside within the bodies of higher-order animals and aid in the digestion of food; others are used for medical purposes such as providing antibiotics. A small subset of microorganisms, however, is harmful. If taken into the body they can cause sickness or even death. As a group, these disease-causing microorganisms are known as *pathogens*.

Pathogens are a serious concern for managers of water resources. Because of the pathogens' small size, they are easily carried by storm water runoff or other discharges into natural waterbodies. Once in a stream, lake, or estuary, they can infect humans through contaminated fish and shellfish, skin contact, or ingestion of water. Of the designated uses listed in section 303(c) of the Clean Water Act, protection from pathogenic contamination is most important for waters designated for recreation (primary and secondary contact); public water supplies; aquifer protection; and protection and propagation of fish, shellfish, and wildlife. Some of the impairments to designated uses caused by pathogens are discussed here.

Recreational use

Excessive amounts of fecal bacteria in surface water used for recreation have been known to indicate an increased risk of pathogen-induced illness to humans. Infection due to pathogen-contaminated recreational waters include gastrointestinal, respiratory, eye, ear, nose, throat, and skin diseases (USEPA, 1986). Gastrointestinal symptoms include vomiting, diarrhea, fever, and stomachache or nausea accompanied by fever. In 1968 criteria were established by the Federal Water Pollution Control Administration (FWPCA) of the Department of the Interior for fecal coliforms at a level of 200 fecal coliform organisms (colony-forming units [CFU[1]] when cultured) per 100 mL of water (USEPA, 1968). In addition to the presence of fecal coliform bacteria in the water column, many studies have shown the presence and survival of fecal coliforms, as well as pathogens, in marine and freshwater sediments (Nix et al., 1994). A study done in Oak Creek, Arizona found that water quality violations only occurred when sediments were found to have high levels of fecal coliforms in the sediments (Crabill et al., 1999). These fecal coliforms may signify the presence of pathogens, which pose a potential health risk. Activities such as recreational swimming that resuspend contaminated sediments and the associated fecal bacteria and pathogens can increase the health risk posed by waters.

In 1986, EPA published *Ambient Water Quality Criteria for Bacteria*-1986. The data supporting the water quality criteria were obtained from a series of research studies conducted by EPA examining the relationship between swimming-associated illness and the microbiological quality of the waters used by recreational bathers (USEPA, 1986).

The results of those studies demonstrated that fecal coliforms, the indicator originally recommended in 1968 by the FWPCA, showed less correlation to swimming-associated gastroenteritis than some other indicator organisms. Two indicator organisms, *E. coli* and enterococci, showed a strong correlation, the former in fresh waters only and the latter in both fresh and marine waters.

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[1]Throughout this document, fecal coliform units are expressed as CFU, counts, organisms, and most probable number (MPN). "CFU" and "MPN" represent units specific to analytical techniques used to quantify fecal coliform concentration, whereas "counts" and "organisms" are generic terms used to express bacteria concentration. In this protocol, specific units (e.g., MPN) are used where appropriate, but all unit expressions are considered equivalent measures of fecal coliform bacteria concentration.

Consequently, EPA's *Ambient Water Quality Criteria for Bacteria*-1986 recommends the use of *E. coli* and enterococci rather than fecal coliforms. The recommended steady-state geometric mean values of these water quality criteria for bacteria are 33 enterococcci per 100 mL and 126 *E. coli* per 100 mL for fresh waters; and a geometric mean of 35 enterococci per 100 mL for marine waters. These values are based on specific levels of risk of acute gastrointestinal illness. The levels of risk used by EPA correlating to these values are no more than eight illnesses per 1,000 swimmers for fresh waters, and no more than 19 illnesses per 1,000 swimmers for marine waters. The illness rates are EPA's best estimates of the accepted illness rates for areas that had previously applied the fecal coliform criterion. EPA determined that when implemented in a conservative manner, these water quality criteria are protective of gastrointestinal illness resulting from primary contact recreation.

Drinking water supply

The presence of any fecal indicators indicates that drinking water is potentially unsafe for consumption. The maximum contaminant level goal (MCLG[2]) is set at zero for *Cryptosporidium* and *Giardia lamblia,* total coliforms, and viruses for public drinking water systems. However, for surface waters used as drinking water, most viruses and bacteria are inactivated by chlorine or other disinfectants used during the treatment process, although some human pathogens are more resistant to disinfection than others. Further, prospective drinking water treatment requirements to reduce the formation of carcinogenic disinfection byproducts during treatment will complicate the task of pathogen control by shifting the use of technology to more advanced and expensive techniques such as ozone, membranes or ultraviolet radiation. All disinfection and filtration technologies are designed to remove a proportion, but not all of, pathogen contamination from the influent *e.g.*, 2 logs or 99 percent removal. Therefore, higher pathogen loadings in the source water (waterbody) translate into higher pathogen contamination levels in the treated water and a greater public risk.

Of particular concern has been the occurrence of the encysted protozoans *Cryptosporidium parvum* and *Giardia lamblia*, which, particularly in the case of *Cryptosporidium*, are not appreciably killed by chlorination and may require special filtration procedures to eliminate risks from exposure to these pathogens. Protozoans can be responsible for causing giardiasis and cryptosporidiosis in humans through ingestion of *Giardia* and *Cryptosporidium* cysts (NCSU, 1997). Giardiasis is a gastrointestinal disease that causes diarrhea and vomiting. Cryptosporidiosis affects the cells of the digestive tract, epithelium, liver, kidneys, and blood. *Cryptosporidium* is capable of causing life-threatening infections in people with weakened immune systems (Graczyk et al., 1998).

Aquatic life and fisheries

Filter-feeding shellfish such as clams, oysters, and mussels, and other shellfish, such as shrimp and crabs, concentrate microbial contaminants in their tissues and may be harmful to humans when consumed raw or undercooked. Fecal and total coliform indicator levels are used to protect consumers of raw bivalve mollusks from viruses causing Norwalk-like viral gastroenteritis, enteric bacteria and Hepatitis A, and the highly pathogenic *Vibrio* bacteria. The Food and Drug Administration (FDA) has established guidelines to reduce the risk from microbial contaminants that might be found in filter-feeding shellfish.

PATHOGEN TYPES

Pathogens most commonly identified and associated with waterborne diseases can be grouped into the three general categories: bacteria, protozoans, and viruses. (Appendix A provides descriptions of the various techniques for measurement of pathogens.)

Bacteria

Bacteria are unicellular organisms that lack an organized nucleus and contain no chlorophyll (Chapra, 1997). They contain a single strand of DNA and typically reproduce by binary fission, during which a single cell divides to form two new cells. Wastes from warm-blooded animals are a source for many types of bacteria found in waterbodies, including the coliform group and *Streptococcus, Lactobacillus, Staphylococcus*, and

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[2] A non-enforceable concentration of a drinking water contaminant that is protective of human health and allows an adequate margin of safety.

Clostridia. Not all bacteria are pathogenic, however. Table 2-1 presents information on some of the major pathogenic waterborne bacteria of concern.

Protozoans

Protozoans are unicellular organisms that reproduce by fission and occur primarily in the aquatic environment. Pathogenic protozoans constitute almost 30 percent (or 10,000) of the 35,000 known species of protozoans (Mitchell et al., 1988, cited in NCSU, 1997). Pathogenic protozoans exist in the environment as cysts that hatch, grow, and multiply after ingestion, manifesting as the associated illness. Encystation of protozoans facilitates their survival, protecting them from harsh conditions such as high temperature and salinity. Two protozoans of major concern as waterborne pathogens are *Giardia lamblia* and *Cryptosporidium*. *Giardia* causes giardiasis, one of the most prevalent waterborne diseases in the United States; *Cryptosporidium* causes cryptosporidiosis. Some waterborne protozoans from fecal sources posing threats to human health are listed with their associated diseases in Table 2-2.

Viruses

Viruses are a group of infectious agents that require a host in which to live. They are composed of highly organized sequences of nucleic acids, either DNA or RNA, depending on the virus. All viruses have a protein covering that encloses the nucleic acid. Some viruses have a lipoprotein (protein in which at least one of the components is a lipid) envelope over the protein covering. The protein or lipoprotein covering determines to what surface the virus will adhere.

The most significant virus group affecting water quality and human health originates in the gastrointestinal tract of infected individuals. These *enteric viruses* are excreted in feces and include hepatitis A, rotaviruses, Norwalk-type viruses, adenoviruses, enteroviruses, and reoviruses. Table 2-3 presents some important viruses and their associated diseases.

Cryptosporidiosis Outbreak in Milwaukee

In 1993, a substantial outbreak of cryptosporidiosis occurred in Milwaukee, Wisconsin. The outbreak was caused by *Cryptosporidium* in the municipally treated drinking water, illustrating the seriousness of the threat of this protozoan in public water supplies. The outbreak is the largest documented waterborne disease outbreak in U.S. history (Craun et al., 1997). At least 26 percent of the population in the five counties constituting the Milwaukee metropolitan area contracted cryptosporidiosis during the 6-week outbreak (Wisconsin Division of Health, cited in Halpern et al., 1997). There were 110 deaths from the outbreak; most of the fatalities were people with weakened immune systems (Milwaukee Health Department, cited in Halpern et al., 1997). According to one study, including the 725,000 lost "work/school days," this outbreak cost an estimated $166 million in medical charges and lost work time (Levin, 1994, cited in Halpern et al., 1997).

Table 2-1. Pathogenic bacteria of concern to water quality and their associated diseases

Bacteria	Disease	Effects
Escherichia coli 0157:H7 (enteropathogenic)	Gastroenteritis	Vomiting, diarrhea
Salmonella typhi	Typhoid fever	High fever, diarrhea, ulceration of the small intestine
Salmonella	Salmonellosis	Diarrhea, dehydration
Shigella	Shigellosis	Bacillary dysentery
Vibrio cholerae	Cholera	Extremely heavy diarrhea, dehydration
Yersinia enterolitica	Yersinosis	Diarrhea

Table 2-2. Protozoans of concern to water quality and their associated diseases

Protozoan	Disease	Effects
Balantidium coli	Balantidiasis	Diarrhea, dysentery
Cryptosporidium	Cryptosporidiosis	Diarrhea, death in susceptible populations
Entamoeba histolytica	Amebiasis (amoebic dysentery)	Prolonged diarrhea with bleeding, abscesses of the liver and small intestine
Giardia lamblia	Giardiasis	Mild to severe diarrhea, nausea, indigestion

Adapted from Metcalf and Eddy, 1991

Table 2-3. Viruses of concern to water quality and their associated diseases[1]

Virus	Disease	Effects
Adenovirus (48 serotypes; types 40 and 41 are of primary concern)	Respiratory disease, gastroenteritis	Various effects
Enterovirus (68 types, e.g., polio, echo, encephalitis, conjunctivitis, and Coxsackie viruses)	Gastroenteritis, heart anomolies, meningitis	Various effects
Hepatitis A	Infectious hepatitis	Jaundice, fever
Reovirus	Gastroenteritis	Vomiting, diarrhea
Rotavirus	Gastroenteritis	Vomiting, diarrhea
Calicivirus (e.g., Norwalk-like and Sapporo-like viruses)	Gastroenteritis	Vomiting, diarrhea
Astrovirus	Gastroenteritis	Vomiting, diarrhea

[1] Hepatitis E is an emerging virus that has caused large outbreaks of infectious hepatitis outside of the U.S.

Adapted from Metcalf and Eddy, 1991 and G. Shay Fout, USEPA, 2000

INDICATOR ORGANISMS

The numbers of pathogenic organisms present in polluted waters are generally few and difficult to identify and isolate, as well as highly varied in their characteristic or type. Therefore, scientists and public health officials typically choose to monitor nonpathogenic bacteria that are usually associated with pathogens transmitted by fecal contamination but are more easily sampled and measured. These associated bacteria are called *indicator organisms*. Indicator organisms are assumed to indicate the presence of human pathogenic organisms. When large fecal coliform populations are present in the water, it is assumed that there is a greater likelihood that pathogens are present (McMurray et al., 1998). Fecal indicators are used to develop water quality criteria to support designated uses, such as primary contact recreation and drinking water supply. EPA publishes 304(a) criteria as guidance to states and tribes. States and tribes may adopt EPA's 304(a) criteria, 304(a) criteria modified to reflect site-specific conditions or criteria based on other scientifically-defensible methods. Fecal indicators may also be used to assess the degree of pathogen removal by treatment processes or to detect contamination of distribution systems.

The selection of fecal indicator organisms is a difficult and controversial process. To function as an indicator of fecal contamination in surface water and groundwater, the organism should (1) be easily detected using simple laboratory tests, (2) generally not be present in unpolluted waters, (3) appear in concentrations that can be correlated with the extent of contamination (Thomann and Mueller, 1987), and (4) have a die-off rate that is not faster than the die-off rate for the pathogens of concern. Some commonly used indicators include coliform bacteria and fecal streptococci. Coliform bacteria, which are able to ferment lactose and produce carbon dioxide gas (CO_2), include total coliforms, fecal coliforms, and *Escherichia coli* (*E. coli*). The term "total coliforms" includes several genera of gram-negative, facultative anaerobic, non-spore-forming, rod-shaped bacteria, some of which occur naturally in the intestinal tract of animals and humans, as well as others that occur naturally in soil and in fresh or marine waters and could be pathogenic to a variety of specific hosts. Fecal coliforms (a subset of total coliforms) include several species of coliform bacteria and are found in the intestines and feces of warm-blooded animals. The presence of *E. coli* (a subset of fecal coliforms) in a water sample also indicates fecal contamination since *E. coli* is one of the ubiquitous coliform members of the intestinal microflora of warm-blooded animals (Jawetz et al., 1987). (For more detailed descriptions of these bacteria, see the glossary.) (See Figure 2-1 for indicator organism relationships.)

There has been a resurgence of interest in the enterococcus group as indicators (Davies-Colley et al., 1994). Enterococci (a subgroup of the fecal streptococci [FS] group) are round, coccoid bacteria that live in the intestinal tract. *Streptococcus faecalis* and *Streptococcus faecium* (part of the enterococci family) are thought to be more human-specific than other streptococci, but they can be found in the intestinal tracts of other warm-blooded animals such as cats, dogs, cows, horses, and sheep. The risk to swimmers of contracting gastrointestinal illness seems to be predicted better by enterococci than by fecal coliform bacteria since the die-off rate of fecal coliform bacteria is much greater than the enterococci die-off rate.

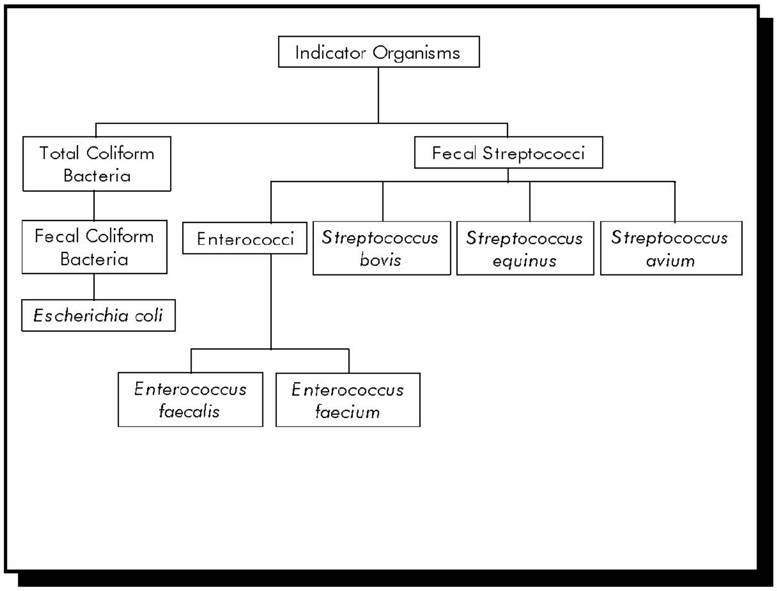


Figure 2-1. Relationships among indicator organisms.

Some officials present fecal coliform and fecal streptococci data as a ratio in an attempt to indicate the origin of bacterial pollution. A fecal coliform/fecal streptococci ratio of 4 or greater has been said to indicate a human source. An FC/FS ratio for domestic animals is on average 0.1-0.6, and the FC/FS ratio for wild animals is on average less than 0.1 (Howel et al., 1995). This generalization, however, does not hold true in many cases (Novotny and Olem, 1994). As applied to FC/FS ratios in surface and ground water samples, these numbers hold true only for recent fecal contamination. The FC/FS ratio is not recommended as a means of differentiating human and animal sources of pollution, mainly because of the variable die-off rates of fecal streptococci species (APHA, 1995).

The 1986 federal bacteriological water quality criteria document (USEPA, 1986) critically reviewed a series of epidemiological and water quality monitoring studies at marine and freshwater beaches since 1972. A comparison of various fecal indicators of potential pathogens with disease incidence revealed that elevated levels of enterococci bacteria were most strongly correlated with gastroenteritis in both fresh and marine recreational waters. The gastroenteritis was assumed to be related to the elevated levels of enterococci. *E. coli* also showed a correlation with gastroenteritis, primarily in freshwater, but total coliform and fecal coliform bacteria, which were commonly measured throughout the United States (established on the basis of the 1968 recommended criteria), were only weakly correlated with this disease. The recommended criteria for enterococci and *E. coli* were then developed. These recommended criteria are discussed in more detail in Section 4 of this document.

Many issues surround the use of fecal indicators in determining the quality of waterbodies relative to pathogens. Major issues of concern are the correlation between the measured indicator and the presence of pathogens and the correlation between those pathogens and the incidence of disease. A review by Pruess (1998) of 22 studies of recreational waters showed that the indicator organisms that correlate best with illness are enterococci/fecal streptococci for both marine water and fresh water and *E. coli* for freshwater. The microbiological indicators yield a general assessment of water quality and safety for the designated or existing use and do not identify specific human pathogens; that is, the exceedance of criteria developed for *E. coli* and enterococci bacteria indicates that the water *might* cause some type of illness following exposure to that water. For example, recreational use by swimmers or surfers could be impaired by the presence of high densities of fecal indicators because there is a chance that some of those microorganisms could cause gastrointestinal illnesses if the water is swallowed. Commercial or recreational harvesting of clams in an estuary could be impaired because the presence of high densities of these bacteria suggests that other human pathogens such as the infectious hepatitis A virus might be present in the shellfish tissues. A public water supply may be impaired by high levels of a pathogen indicator originating from human sources or activities.

PATHOGEN SOURCES AND TRANSPORT

Pathogenic organisms are one of many types of pollutants generated at a source (point or nonpoint) and then transported by a pipe, storm water runoff, groundwater, or other mechanism to a body of water. Identifying these sources and tracking the movement of pathogens is often a difficult and resource-intensive task.

Point sources

The transport of pathogens to a waterbody occurs either directly or indirectly from both point and nonpoint sources. For point sources, the direct transport pathway is straightforward—the point source (e.g., wastewater treatment plant [WWTP]) end-of-pipe pathogen concentration is directly discharged into a waterbody. Major point sources of pathogens are discharges from WWTPs and combined sewer overflows (CSOs). Raw sewage entering the WWTP typically has a total coliform count of 10^7 to 10^9 most probable number (MPN) per 100 mL (Novotny et al., 1989). Associated with raw sewage are proportionally high concentrations of pathogenic bacteria, viruses, and protozoans. A typical plant reduces the total coliform count by about 3 orders of magnitude, to the range of 10^4 to 10^6 MPN/100 mL. The magnitude of pathogen reduction, however, varies with the treatment process employed. For example, the protozoan *Giardia* is treated effectively with chlorine, but chlorine does not effectively kill *Cryptosporidium* (Chapra, 1997). For *Cryptosporidium*, filtration or ozonation must be applied.

In some instances raw sewage can bypass WWTPs and enter waterbodies directly. This can occur because of failures or leaks in sanitary sewer systems or, in the case of CSOs, when wet-weather flows exceed the conveyance and storage capacity of the combined system. In CSOs, urban runoff and sanitary sewage are conveyed in the same system. Typical CSO concentrations for total coliforms are reported as 10^5 to 10^7 MPN/100 mL (Novotny et al., 1989), or about 1 order of magnitude greater than treatment plant effluent. In contrast to WWTP effluent, CSOs discharge for short periods of time, discharge at random intervals, and are associated with storm flows that provide dilution of the effluent.

Other point sources that can contribute substantial loads of pathogens and fecal indicators to waterbodies include concentrated animal feeding operations, slaughterhouses and meat processing facilities; tanning, textile, and pulp and paper factories; and fish and shellfish processing facilities.

Nonpoint sources

Nonpoint sources of pollution differ from point sources because the former are typically wet-weather-dominated. In addition, nonpoint source pollutants are diffuse in nature and do not enter waterbodies from any single point. Indirect nonpoint sources include any source located far enough from waterbodies to allow attenuation of the pathogens in runoff, infiltrated water, or groundwater. Identification of sources and quantification of pathogen loads from nonpoint sources can be difficult. In urban[3] and suburban areas, nonpoint sources of pathogens include urban litter, contaminated refuse, domestic pet and wildlife excrement, and failing sewer lines. In a study of bacterial loading in urban streams, Young and Thackston (1999) found that fecal bacteria densities were directly related to the density of housing, population, development, percent impervious area, and domestic animal density.

Rural nonpoint source loads originate from both land use-specific and natural sources. The primary rural nonpoint source for pathogens is confined animal operations, in which large quantities of fecal matter are produced. Livestock excrement from barnyards, pastures, rangelands, feedlots[4], and uncontrolled manure storage areas is a significant nonpoint source of bacteria, viruses, and protozoal cysts. The occurrence and degree of fecal indicator and pathogen loads from livestock are linked to temporally and spatially variable hydrologic factors such as rainfall and runoff except when manure is deposited directly into a waterbody (Edwards et al., 1997). Other significant sources include leaking septic systems and land application of manure and sewage sludge[5]. Septic systems that fail hydraulically (surface breakouts) or hydrogeologically (inadequate soils to filter pathogens) can adversely affect downgradient surface waters (Horsley and Witten, 1996). Because the majority of pathogens are filtered or attenuated in soil zones over

[3] Some urban stormwater sources are considered as point sources by the CWA.

[4] If feedlots meet the regulatory definition of a Concentrated Animal Feeding Operation (CAFO), they are treated as point sources by the CWA and therefore are not considered in nonpoint source load contributions.

[5] Much of the application of sewage sludge is regulated by permits under state and federal laws.

the water table, groundwater has traditionally been considered the water source least susceptible to contamination by pathogens. However, depending on soils and geology, connections between groundwater and a contaminated surface or subsurface source can pose threats to the quality of aquifers in the area. Seepage from a waste lagoon, a leaking septic tank, or an improperly designed landfill can result in contamination of aquifer resources.

Wildlife can also be a significant nonpoint source of pathogens in many areas. Many wildlife species are reservoirs of microorganisms that are potentially pathogenic to themselves and to humans. Beaver and deer are large contributors of *Giardia* and *Cryptosporidium*, respectively. Waterfowl such as geese, ducks, and heron also can contaminate surface water with microbial pathogens (Graczyk et al., 1998). These pathogens, such as *Giardia* cysts, are a potentially dangerous health risk for humans, livestock, and wildlife.

Although many nonpoint sources of pathogens are diffuse in nature, some can act as direct sources to a waterbody. Examples of these direct nonpoint sources of pathogens are boat discharges, landfills, waterfowl, and failing septic systems. Boats lacking holding tanks for pumpout contribute human pathogens to surface water; groundwater impacts could occur due to seepage from landfill oxidation ponds that contain fecal bacteria (Metcalf and Eddy, 1991); waterfowl contributions of pathogens are often directly deposited to the waterbody of concern; and failing septic systems may contribute significant pathogen loads directly to a waterbody without significant reduction in numbers, especially in coastal areas or areas of coarse-textured soils or karst geology.

Another potential nonpoint source of pathogens is the resuspension of bacteria indicators and pathogens in sediments. For example, Weiskel et al. (1996) reported significantly increased values of water column fecal coliform density after artificial disturbance of the surface 2 cm of sediments in Buttermilk Bay, Massachusetts. These increased levels of fecal coliform bacteria might indicate the presence of pathogens in the waterbody. The most pronounced increases occurred at sites underlain by fine-grained, high-organic-carbon muds. As runoff during a storm event begins, the discharge and velocity increase, in turn scouring bacteria from the benthic areas of the stream (Yagow and Shanholtz, 1998). This scouring causes increased levels of bacteria concentrations in the water column and decreased levels in the stream sediments. After peak discharge, the bacteria concentrations in the water column decrease at a faster rate than the discharge. This causes the sediment to be deposited downstream, where the sediment bacteria concentrations increase and water column concentrations return to background levels. The increasing usage of recreational waters can cause resuspension of the high numbers of bacterial indicators and pathogens occurring in the sediments (Burton et al., 1987). This creates a potential health hazard from the possible ingestion of the resuspended pathogens.

Although the type of source provides information on the concentrations and possible loads of pathogens to waterbodies, another important consideration is the proximity of the source to the waterbody of concern. Nonpoint sources closer to a waterbody have a greater likelihood to pollute the water than those located farther away, where attenuation factors and dilution will reduce the actual load delivered to the waterbody.

FACTORS INFLUENCING PATHOGEN SURVIVAL

Determining what happens to the microorganisms once they reach the waterbody is often as challenging as identifying and tracking their sources. As living organisms they require certain conditions to survive, grow, and reproduce. Thus, risks to human health can be increased or decreased depending on water temperature and other factors associated with the waterbody. Many factors influence the die-off rate of viruses, bacteria, and protozoans in the environment. These factors include sunlight, temperature, moisture conditions, salinity, soil conditions, waterbody conditions, settling, association with particles, and encystation. Many other factors affect the die-off rate of pathogens, but not all are described in this protocol. Some of these other factors include the age of the fecal deposit, pH, starvation, structural damage, chemical damage, predation (Davies-Colley et al., 1994), osmotic stress in moving from fresh to marine waters, nutrient deficiencies, turbidity (water clarity), variation of spectral quality of sunlight, microbial composition of effluents, and oxygen concentrations. Some of these factors have a direct influence on mortality, whereas others indirectly affect die-off in the environment by increasing exposure to other factors. For example, a longer distance traveled from the source to the waterbody affects die-off by increasing exposure to attenuation factors, such as temperature, sunlight, and moisture. The factors are influenced by many variables, the most notable being the medium in which they occur. For example, ultraviolet light increases the die-off rate of fecal indicator bacteria, but the magnitude of the die-off is different if the bacteria are on the ground surface, in the upper water column, or in the lower water column.

Although die-off rates vary by species for each group of pathogens and bacterial indicators, the following overview describes the general factors that influence pathogen extinction; organism- or species-specific effects are not discussed. Of the many direct factors that might influence the inactivation of pathogens in the environment, the most important are sunlight (ultraviolet and near ultraviolet radiation), temperature, and moisture conditions. In addition, many other factors potentially influence pathogen mortality in the environment. Some factors may affect die-off by prolonging direct exposure to the attenuation factors listed above; others, such as predation, affect die-off but are less important than sunlight, temperature, or moisture.

Sunlight (ultraviolet radiation)

Bacterial survival after deposition onto the land surface is greatly dependent on solar radiation, especially in the ultraviolet range (Auer and Niehaus, 1992). Because of solar radiation, bacteria have a shorter survival time at the surface than in soil. Solar radiation is also a major factor in the survival time of viruses. Increased solar and ultraviolet radiation greatly decreases the survival rate of viruses, and like bacteria, viruses have a decreased survival time on the surface relative to survival in soil. Limited information is available on the influence of ultraviolet radiation on protozoans. Sunlight does, however, play an important role in the inactivation of *Giardia* and *Cryptosporidium* (Johnson et al., 1997). In a study done by Johnson et al. (1997), *Giardia* cyst and *Cryptosporidium* oocyst die-off rates were both affected by sunlight. Both protozoans persisted longer in the dark than in direct sunlight, but *Cryptosporidium* oocysts survived longer. Johnson et al. (1997) found that the order of survival for some waterborne pathogens in sunlight to be *Cryptosporidium* > poliovirus > *Giardia* > *Salmonella*.

Temperature

Pathogen and bacterial indicator survival is highly dependent on temperature. Temperature has an inverse relationship with the survival of microorganisms originating in fecal waste, with survival decreasing as temperature increases. Many laboratory studies have been conducted to determine conditions that affect the infectivity of *Cryptosporidium* spp. oocysts. Researchers found infectivity was lost when the oocysts were frozen, boiled, or heated to 60 °C or more for 5 to 10 minutes or longer (Badenoch et al., 1990); freeze-dried (Tzipori, 1983); or stored for 2 weeks at 15 to 20 °C or stored for 5 days at 37 °C (Sherwood et al., 1982). Oocysts of *Cryptosporidium* have been observed to survive for up to 6 months in river water at ambient temperatures (Medema et al., 1997). *Giardia lamblia* cysts can survive more than 2 months at 4 °C (Adam, 1991; Bingham et al., 1979). Temperature is apparently the major factor for virus and coliform bacteria survival in soils, with an estimated doubling of the die-off rate for each 10 °C rise (Gerba and Bitton, 1984; Reddy et al., 1981). Temperature is also the dominant factor affecting virus survival in freshwater, with greater survival occurring at lower temperatures. Enteric viruses can survive from 2 to more than 188 days in freshwater (Novotny and Olem, 1994).

Moisture

Soil moisture is another important factor in the survival time of bacteria in soil. Survival time of bacteria increases with the moisture content and moisture holding capacity of the soils (Reddy et al., 1981). Typically, higher clay content in soil results in increased soil moisture retention and, consequently, increased bacteria survival. *Cryptosporidium* oocysts lost their infectivity when dried for 1 to 4 days at -1 to 29 °C (Anderson, 1986). Dry fecal specimens lost their infectivity more rapidly than those kept moist.

Salinity

The survival of bacteria in water is largely dependent on salinity. Chapra (1997) reports a formula for the calculation of the natural mortality rate of total coliforms that assumes a freshwater loss rate of 0.8 organisms per day (d^{-1}) regardless of outside factors. The freshwater loss is supplemented by a saltwater loss that is linearly dependent on salinity, resulting in a total loss rate range of 0.8 d^{-1} for freshwater to 1.4 d^{-1} for saltwater. The total loss can be modified to account for other factors, such as temperature or insolation.

Soil conditions

Pathogen survival in soil is affected by such soil conditions as pH and predation. Shorter survival times have been noted in acid soils (pH 3 to 5) than in neutral calcareous soils (Novotny and Olem, 1994). In general, bacteria survival decreases in soil with low pH, with bacteria attenuation occurring in the soils with pH levels between 3 and 4 (Horsley and Witten, 1996). Viruses cannot reproduce in soil, but can survive in soil from as short a time as 7 days to as long as 6 months, depending on the nature of the soil, temperature, pH, moisture, and predation by soil microflora (Howell et al., 1996, Novotny and Olem, 1994). Longer survival of some bacteria and viruses has also been noted when sufficient amounts of organic matter are present.

Waterbody conditions

After discharge into a waterbody, pathogenic organisms are subject to many additional factors during dispersion and transport. The factors that influence the survival of the pathogenic organisms within the waterbody are the physical conditions of the water (Baudisova,1997), sunlight, temperature, salinity, predation, nutrient deficiencies, toxic substances, settling, resuspension of particles with sorbed organisms, and aftergrowth (growth of the organisms in the waterbody) (Thomann and Mueller, 1987). Typically, conditions favorable to the survival of pathogens in water are lower amounts of light energy, lower salinity, elevated levels of nutrients and organic matter, and lower temperatures.

Settling

Many studies have shown that there are often much higher numbers of indicator and pathogenic bacteria in sediments than in the overlying waters (Burton et al., 1987). These higher concentrations of bacteria in the sediments are apparently due to a combination of sedimentation, sorption, and the phenomenon of extended survival in sediments. Bacterial cells settle from the water column as discrete entities and as part of larger aggregates of fecal material, storm water debris, and other suspended solids (Schillinger and Gannon, 1982, cited in Auer and Niehaus, 1992). Gannon et al. (1983) concluded that sedimentation played an important role in the overall removal of fecal coliform from the water column after observing that viable fecal coliform bacteria accumulated at the sediment surface in Ford Lake, Michigan. Once settled, pathogens and bacterial indicators can have an increased survival time due to protection from harmful factors such as sunlight and temperature. Levels of fecal coliform and specific pathogenic organisms have been shown to survive for longer periods of time in the sediments than in the overlying water column (Sherer et al., 1992; Burton et al., 1987; Thomann and Mueller, 1987). The sediment reservoir allows for the enteric and pathogenic bacteria to survive for up to several months, making resuspension and ingestion in primary contact waters a real threat to swimmers (Burton et al., 1987). Increased survival rates for viruses in estuarine sediments have been reported in LaBelle and Gerba (1980), Roper and Marshall (1979), Burton et al., 1987 and Sherer et al., 1992. Due to the accumulation of pathogens in bottom sediments, resuspension of the sediment and the subsequent desorption of the pathogens is a potential source of contamination to the overlying water. A study by Sherer et al. (1992) showed the survival of fecal coliform and fecal streptococci to be significantly longer in sediment-laden waters than in waters without sediment. Fecal coliform and fecal streptococci bacteria showed half-lives from 11 to 30 days and 9 to 17 days, respectively, when incubated with sediment. These are longer half-lives than those when sediment was not present. During the study the stream bottom was disturbed several times. The mean concentration of fecal coliform in the stream increased by 1.7 times the initial concentration after the stream bottom was disturbed. The fecal streptococci concentration increased by 2.7 times. This study showed that enteric bacteria can survive in sediments for several months as compared to a only few days in the overlying water.

Encystation

Protozoans occur primarily in aquatic environments, where they exist in resting stages, called cysts or oocysts. *Giardia* cysts can survive in water for 1 to 3 months (NCSU, 1997). Although protozoans can extend their survival time by encystation, the cysts and oocysts can become nonviable in the environment, causing only a fraction of the total concentration to be capable of leading to infection. Lower viability tends to occur at high temperatures (Chapra, 1997).

PATHOGEN SOURCE CONTROLS

A key objective of water quality protection is to protect human health from the deleterious effects of waterborne pathogens. Water quality standards define the goals for a waterbody by designating the use(s), by setting numeric or narrative criteria necessary to protect the use(s), and by protecting water quality through antidegradation provisions. The numeric or narrative criteria are to be based on sound scientific rationale and should contain sufficient parameters or constituents to protect the designated or existing use.

Controlling point sources

NPDES permits are required for the discharge of pollutants from most point source dischargers into the waters of the United States. These permits translate wasteload allocations into enforceable limits and requirements for point sources by setting restrictions on the quantities, discharge rates, and/or concentrations of the specified pollutants. Point sources typically rely on a range of treatment options before discharging effluent. Treatment of municipal waste is generally identified as primary, secondary, or advanced (previously called *tertiary* treatment), although the distinctions are somewhat arbitrary. Primary treatment involves removing suspended solids with screens and the use of gravity settling ponds followed by disinfection. Most protozoan cysts settle out in ponds after 11 days due to their size (*Environmental Microbiology,* 1997).

Secondary treatment uses biological treatment to decompose organic matter to cell material and by-products, and the subsequent removal of cell matter, usually by gravity settling. Secondary treatment can also be followed by disinfection. Activated sludge processes involve the production of an activated mass of microorganisms capable of stabilizing waste aerobically. Aerobic processes are preferred due to their higher rates of decomposition and because pathogenic microorganisms tend to grow poorly or not at all under aerobic conditions. Secondary treatment by activated sludge typically reduces coliform bacteria concentrations by 90 to 99 percent.

Advanced treatment is any practice beyond secondary treatment and is very effective in destroying most pathogens. Advanced treatment can include filtration, coagulants, and disinfection. Conventional filtration units are helpful prior to disinfection in removing substances that interfere with effluent disinfection (Wright, 1997). An emerging practice is the use of microfiltration after pretreatment (primary), during which water passes through clusters of 20,000 fibers with a nominal pore size of 0.2 micron (Wright, 1997). These microfilters easily capture *Cryptosporidium* oocysts (3 to 7 microns in diameter). Conventional filtration units are aimed at removing larger particles of 15 to 30 microns. Chemical pretreatment involves the addition of alum or other chemicals to form clumps of impurities, or floc, which settle out or are easily filtered out of the raw drinking water.

Disinfection is the most common treatment technique to combat waterborne diseases, and can be used as part of primary, secondary, and advanced treatment. The most frequently used disinfectant is chlorine, which kills many microbes, including most pathogens, except encysted protozoans, which are resistant to chlorine (Bryant et al., 1992). However, chlorine's efficiency is a function of initial mixing, contact time, temperature, pH, amount of residual, and characteristics of the microorganisms (such as their age). Application of chlorine in a highly turbulent system will result in kills 2 orders of magnitude greater than those when chlorine is added separately to a complete-mix reactor with constant and uniform distribution (Metcalf and Eddy, 1991). Different chlorine compounds are used for disinfection, with chlorine dioxide being equal to or greater than chlorine in disinfecting power. Chlorine dioxide has been proven to be more effective than chlorine in the inactivation of viruses, but produces the toxic and problematic byproduct of chlorite. Chloramine might also be more effective than chlorine because it breaks down slowly, resulting in longer-lasting disinfection properties. All chlorine disinfection is dependent on concentration of the chlorine residual and temperature of the water, and time of contact.

Other disinfectants used are ozone, ultraviolet light, and iodine. Ozone is an extremely reactive oxidant that kills pathogens directly through cell wall disintegration. Ozone is believed to be more effective in killing viruses than are chlorine compounds (Metcalf and Eddy, 1991), and it is thought to be an effective means of eliminating *Cryptosporidium* (Oppenheimer *et al.*, 2000; Wright, 1997). Ultraviolet (UV) light penetrates the cell wall of microorganisms and is absorbed by cellular material, which either prevents replication or causes death of the cell. Disinfection by UV light is more effective at shallower depths and lower turbidity because turbidity absorbs the UV energy and shields the pathogens. UV light does not leave a residue in the water to kill remaining organisms during discharge and is not effective against *Giardia*.

The most recommended and effective approach to removal of pathogens is a multiple-barrier approach using some combination of sedimentation, chemical pretreatment and flocculation, filtration, and disinfection. Complete water treatment with chemical coagulation, filtration, and disinfection might be necessary to effectively treat encysted protozoans.

Controlling nonpoint sources

The use of best management practices (BMPs) should consider the most efficient and cost-effective methods to achieve load allocations for nonpoint sources. Because livestock operations contribute high pathogen loads, agricultural BMPs may provide considerable reduction of rural nonpoint source pollution by pathogens. Methods to control agricultural nonpoint sources include minimizing the source, minimizing the movement (to increase die-off), and treating the water. BMPs can be classified into three categories—management, structural, and vegetative. To select the most effective BMP or combination of BMPs, a manager must determine the primary source of the pollutant and its method of transport to the waterbody. Some controls associated with BMPs are listed in Table 2-4.

PATHOGEN TMDLS

This protocol provides a step-by-step description of the TMDL development process for pathogens and includes case studies and hypothetical examples to illustrate the major points in the process. The protocol emphasizes the use of rational, science-based methods and tools for each step of TMDL development. TMDL development is site-specific. The availability of data influences the types of methods that developers can use. Ideally, extensive monitoring data are available to establish baseline water quality conditions, pollutant source loading, and waterbody system dynamics. However, without long-term monitoring data, the developer will have to use a combination of monitoring, analytical tools (including models), and qualitative assessments to collect information, assess system processes and responses, and make decisions. Although some aspects of TMDLs must be quantified (e.g., numeric targets, loading capacity, and allocations), qualitative assessments are acceptable as long as they are supported by sound scientific justification or result from rigorous modeling techniques. A goal of this document is to assist developers in using a rational TMDL development process that incorporates the required elements of a TMDL.

Range of viable TMDL approaches

Analysts should be resourceful and creative in selecting TMDL approaches and should learn from the results of similar analytical efforts. The degree of analysis required for each of the components of TMDL development can range from simple, screening-level approaches based on

Table 2-4. Methods of control for agricultural nonpoint sources and their associated types of controls

	Types of Controls		
Methods of Control	**Structural**	**Vegetative**	**Management**
Minimize source	Fences (livestock exclusion)		Animal waste management, especially proper application rate and timing
Minimize movement	Animal waste storage; detention pond	Filter strips; riparian buffer zones	Proper site selection for animal feeding facility; proper waste application rate
Treat water	Waste treatment lagoon; filtration	Artificial wetland; rock reed microbial filter	Recycle and reuse

Source: Novotny and Olem, 1994.

limited data to detailed investigations that might take several months or even years to complete. A variety of interrelated factors affect the degree of analysis necessary. These factors include the type of impairment (e.g., violation of a numeric criterion versus designated or existing use impairment); the physical, chemical, and biological processes occurring in the waterbody and its watershed; the size of the watershed; the number of sources; the data and resources available to develop the TMDL; and the types and costs of actions needed to implement the TMDL (Figure 2-2).

Decisions regarding the extent of the analysis must always be made on a site-specific basis as part of a comprehensive problem-solving approach. TMDLs are essentially a problem-solving process to which no "cookbook" approach can be applied. Not only will analyses for different TMDL studies vary in complexity, but the degree of complexity in the methods used within individual TMDLs might also vary substantially. Screening-level approaches afford cost and time savings, can be applied by a wide range of personnel, and are generally easier to understand than more detailed analyses.

The trade-offs associated with using simpler approaches include a potential decrease in predictive accuracy and often an inability to predict water quality at fine geographic and time scales (e.g., watershed-scale source predictions versus parcel-by-parcel predictions, and annual estimates versus seasonal estimates). When using simpler approaches, analysts should consider these two shortcomings in determining an appropriate margin of safety.

The advantages of more detailed approaches are presumably an increase in predictive accuracy and greater spatial and temporal resolution. These advantages can translate into greater stakeholder acceptance and a smaller margin of safety, which usually reduces source management costs. Detailed approaches might be necessary when the screening-level approaches have been tried and have proven ineffective or when it is especially important to "get it right the first time" (e.g., where protection from waterborne diseases is a TMDL issue). In addition, more detailed approaches might be warranted when there is significant uncertainty regarding whether pathogen discharges are attributable to human or to natural sources and the

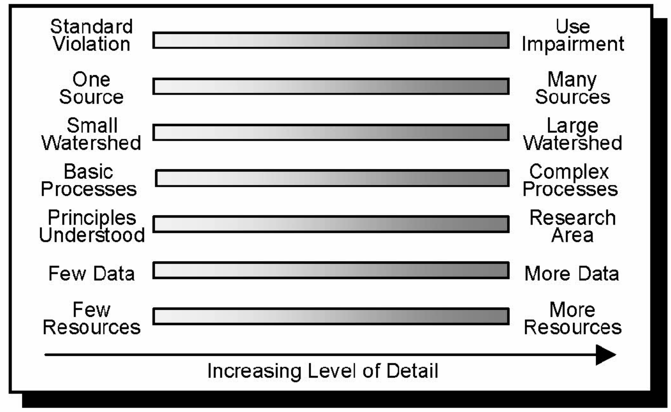


Figure 2-2. Factors influencing the level of detail for the TMDL analysis

anticipated cost of controls is especially high. However, more detailed approaches are likely to cost more, require more data, and take more time to complete.

A variety of approaches to developing a TMDL are justifiable as long as they adequately identify the load reductions or other actions needed to restore designated or existing uses. Because all situations requiring development of a TMDL are different, one cannot specify that if X and Y are true a certain approach must be used. Site-specific factors should always be taken into account and an appropriate balance struck between cost and time issues and the benefits of additional analyses.

PATHOGEN TMDL EXAMPLES

The following brief summaries of five pathogen TMDLs show that a range of methods is appropriate for TMDL development and that individual TMDLs often combine relatively detailed analysis for certain elements with simple analysis supporting other elements. A more detailed case study is provided in Appendix B.

Republican River, Kansas

Fecal bacteria contamination had been identified in two segments of the main stem of the Republican River and two tributary segments (Crosby and Otter creeks) in Kansas (KDHE, 1999). The main stem segments are designated for primary and secondary recreation, aquatic life support, domestic water supply, food procurement, and irrigation/stockwater. The designated uses for the two tributary segments are aquatic life support and secondary contact recreation. Elevated fecal coliform bacteria loadings are mainly from nonpoint sources in the watershed. These elevated fecal coliform levels are causing impairment of primary and secondary contact recreation use on the main stem and secondary contact recreation on Crosby and Otter creeks. Placement on Kansas' 303(d) list was supported by in-stream monitoring that indicated that 10 percent of spring samples and 41 percent of summer-fall samples exceeded the primary criterion for fecal coliform bacteria. Overall, 17 percent of the samples exceeded the water quality criteria.

Development of a fecal coliform bacteria TMDL for the Republican River began with an assessment of the existing fecal coliform loads to the river. Nonpoint sources of fecal coliform loading to the river include livestock waste management systems, runoff from cropland and grassland, and wildlife (although loading from wildlife is minimal). There are no point sources in this watershed. To determine the needed load reductions Kansas Department of Health and Environment (KDHE) used a TMDL curve methodology. The TMDL curve is the concentration of fecal coliform bacteria per day vs. the percent of days the load is exceeded at a specific monitoring station. Points falling above the curve represent deviations from the water quality standard and the permissible loading function. Points falling below the curve represent compliance with standards and support for the designated use. The curve helps to determine the issues surrounding the problem and differentiate between point and nonpoint sources; show seasonal water quality effects; address frequency of deviation, magnitude of deviation, and duration questions; compare water quality conditions between multiple watersheds; and establish the level of implementation needed. Loads that fall above the curve in the flow regime defined as being exceeded 85-99 percent of the time are considered point source influences. Points that fall above the curve over the range of 10-70 percent exceedance are considered to be nonpoint sources. Therefore, the percentage of the area to the right of the 85 percent exceedance mark is the Wasteload Allocation and percentage of the area to the left of the 85 percent exceedance mark is the Load Allocation.

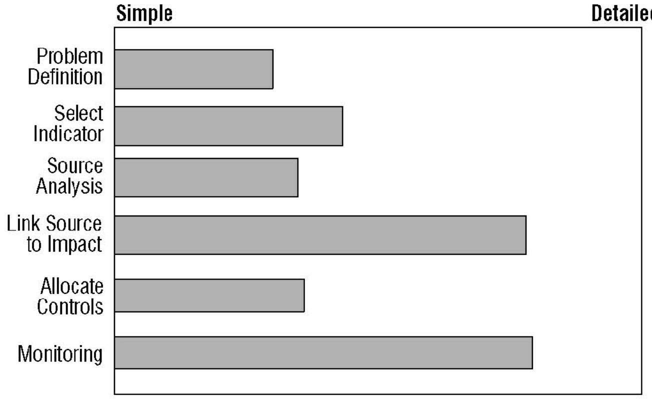
The nature of bacteria loading is too dynamic to assign fixed allocations for wasteloads and nonpoint loads. Instead, allocation decisions were made that reflect the expected reduction of bacteria loading under defined flow conditions. There are no point sources in the watershed, and therefore wasteload allocations established under this TMDL are equal to zero. The proposed allocation plan requires that less than 10 percent of samples taken in spring exceed the primary criterion at flows under 660 cubic feet per second (ft^3/s), with no samples exceeding the criterion at flows less than 165 ft^3/s; less than 10 percent of samples taken in summer or fall exceed the primary criterion at flows under 660 ft^3/s, with no samples exceeding the criterion at flows less than 140 ft^3/s; and less than 10 percent of samples taken in winter exceed the secondary criterion at flows under 660 ft^3/s. These endpoints will be reached through unspecified reductions in loading from the smaller, unpermitted livestock operations and rural homesteads and farmsteads in the watershed. Best management practices will be directed toward those activities in the upstream watersheds so that there will be accrued benefits of reduced violations of the applicable fecal coliform criteria at higher flows on the main stem of the river.

To determine whether the TMDL will improve conditions to support designated uses and maintain water quality standards, KDHE will continue to collect bimonthly samples during the spring, summer-fall, and winter from 1999 to 2003. The status of the 303(d) listing will be evaluated in 2004 based on these samples. If the impaired status remains in 2004, the desired endpoints will be refined and more intensive sampling will be conducted under specified seasonal flow conditions from 2004 to 2008.

Lower Geddes Pond, Michigan (Preliminary)

This preliminary example is based on a study conducted on Lower Geddes Pond, Michigan. The pond is a segment of the Huron River near Ann Arbor, Michigan. The results of the study have not yet been used to prepare a TMDL submittal, but they have been used to discuss the options for TMDL development. The Lower Geddes Pond example has been included despite its preliminary nature because of the use of *E. coli* as the indicator bacteria for this waterbody.

Republican River



Water Quality Indicators	TMDL	Controls
Primary Criterion for Fecal Coliform % exceedance dependent on season and flow conditions	WLA and LA derived from TMDL Curve Methodology	Reduction in loads from small, unpermitted livestock operations and rural homesteads and farmsteads
Source Analysis: Qualitative Source Characterization Link to Indicator: TMDL Curve Methodology		

Lower Geddes Pond has been placed on Michigan's 303(d) list because of impairment of total body contact recreational uses by elevated levels of pathogens, and it requires the development of a TMDL for the indicator bacteria *E. coli* (LTI, 1999). The water quality standards in the state of Michigan for *E. coli* require that all waters of the state protected for total body contact recreation may not exceed 130 *E. coli*/100 mL, as a 30-day geometric mean, and at no time may the waters of the state protected for total body contact recreation exceed a maximum of 300 *E. coli*/100 mL. Current data for *E. coli* levels in Lower Geddes Pond are not available, but data are available for fecal coliform bacteria. Linear regression was used to estimate the levels of *E. coli* in Lower Geddes Pond based on fecal coliform levels (Figures 2-3 and 2-4.) This relationship seems possible for wet weather, but dry weather shows more variability. The linear regression shows an estimated *E. coli* geometric mean for wet and dry weather well below the state standard. The estimated *E. coli* maximum, however, is well above the state standard of 300/100 mL.

The available data are sufficient to verify the existence of a problem, but are not sufficient to provide detail on the sources contributing bacteria to the waterbody. Additional sampling can be conducted to help identify the existing sources of bacteria to Lower Geddes Pond. Techniques such as DNA fingerprinting can also be used to help in identifying whether the bacteria are of human, wildlife, or domestic pet origin. Detailed sampling can be conducted throughout the watershed to determine the

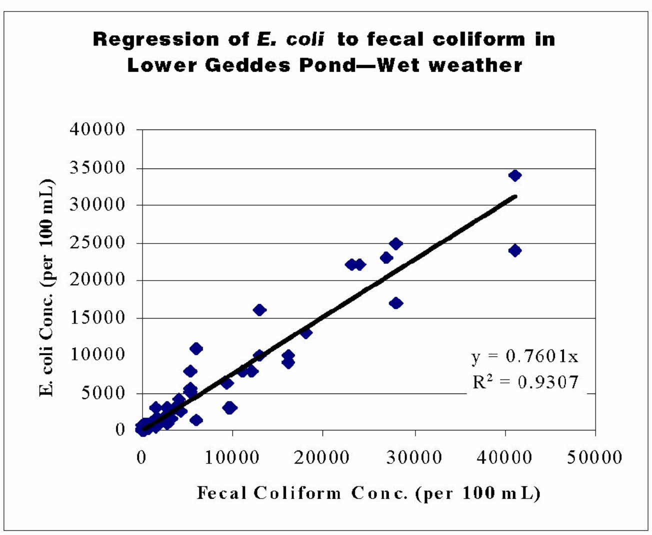


Figure 2-3. Regression of *E. coli* to fecal coliform for Lower Geddes Pond samples—wet weather

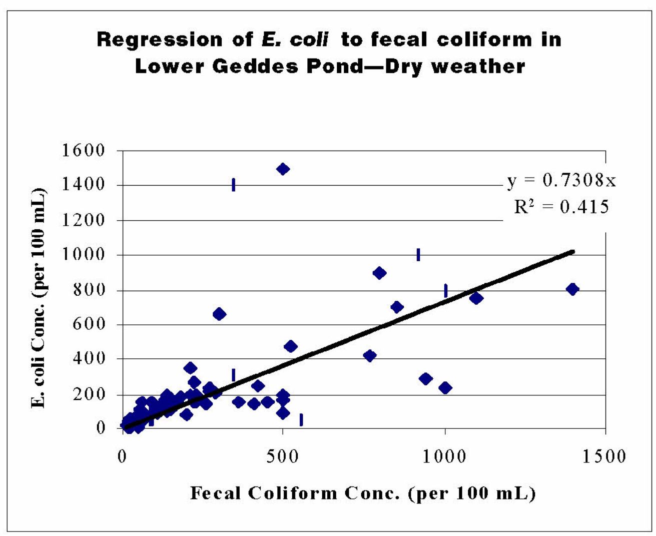
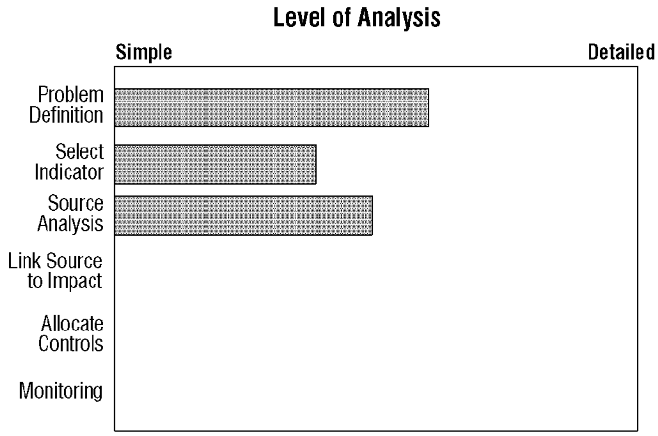


Figure 2-4. Regression of *E. coli* to fecal coliform for Lower Geddes Pond samples—dry weather

Lower Geddes Pond



Water Quality Indicators	TMDL	Controls
130 E. Coli/ 100 mL (30-day geometric mean) 300 E. Coli/ 100 mL maximum	Not Applicable	Not Applicable
Source Analysis: Qualitative Source Characterization Link to Indicator: Not Applicable		

contributions and impacts of subwatersheds and individual land use categories.

Additional monitoring will be required to identify the sources of bacteria to Lower Geddes Pond and to confirm that water quality standards are being met. Without further monitoring, bacteria conditions in Lower Geddes Pond will remain highly uncertain and the success of implementation efforts will be unknown.

There are two possible alternatives for the development of the Lower Geddes Pond TMDL. The first option is to conduct a phased TMDL using the existing data. The second option is to conduct extensive additional sampling before TMDL development. The first option can generate an approvable TMDL in a shorter amount of time, but cannot include an implementation plan because of the lack of current data. Adjustments would have to be made to the TMDL as new data are collected and analyzed. The second option would postpone development of the TMDL until suitable data are collected. The main difference between these two approaches is the timing of the different elements.

Rio Chamita, New Mexico

Rio Chamita, New Mexico, flows from its headwaters in Colorado to its connection with the Rio Chama below the village of Chama, New Mexico (TMDL for the Rio Chamita, undated). The Rio Chamita is within the 38-mi^2 Rio Chama Basin. Part of the river is located within the Edward Sargent Fish and Wildlife Area. The river has several significant tributaries and groundwater inputs. Eighty-five percent of the surrounding land is in New Mexico, while 15 percent of the surrounding land belongs to the state of Colorado. Land uses in the state of New Mexico include rangeland (42 percent), forest (43 percent), and water (<1 percent). The designated uses for the river include high-quality coldwater fishery, domestic water supply, fish culture, irrigation, livestock watering, wildlife habitat, and secondary contact recreation. The Rio Chamita was placed on the New Mexico 303(d) list with fecal coliform as a pollutant of concern. Elevated fecal coliform levels have impaired the designated use of the river as a high-quality coldwater fishery.

The Rio Chamita's standards require that the monthly geometric mean of fecal coliform bacteria may not exceed 100 fcu (fecal coliform units)/100 mL and no single sample may exceed 200 fcu/100 mL. Two significant sources of fecal coliform bacteria have been identified for this segment of the Rio Chamita. One source is the Village of Chama WWTP, which is a point source. The Village of Chama WWTP serves a population of about 400 people and is monitored through an NPDES permit. The current permit allows a 7-day geometric mean fecal coliform limit of 500 fcu/100 mL and a 30-day geometric mean of 500 fcu/100 mL. These limits are not consistently met. Uncharacterized nonpoint sources of fecal coliform also cause fecal coliform levels upstream of the WWTP discharge to be above current stream criteria. Current fecal coliform levels in the river from nonpoint sources average 450 fcu/100 mL, which is well above the allowable amount of 100 fcu/100 mL.

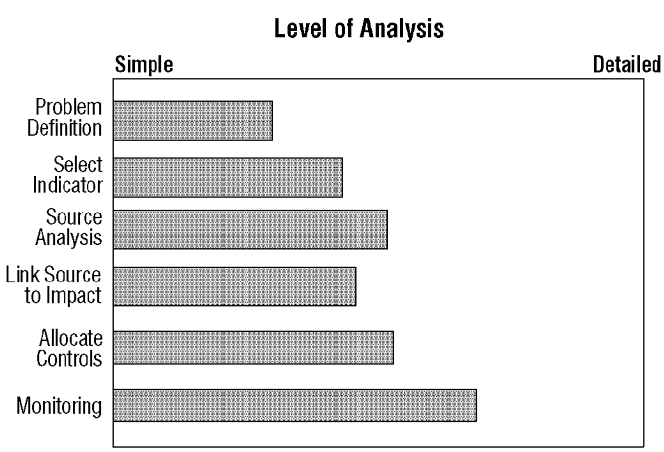
Using the 4Q3 low flow and the target concentration of 100 fcu/100 mL, loading capacity of the stream has been calculated to be 1.117 x 10^{10} fcu/day. The NPDES permit for the WWTP had a limit that was five times the applicable water qaulity criterion and frequently discharged in exceedance of their permit limits and the Rio Chamita water quality standards for fecal coliform bacteria. The end-of-pipe discharge was lowered to equal the in-stream water quality standard. Using the limit of 100 fcu/100 mL and the WWTP design flow, a wasteload allocation for the WWTP has been set at 1.136 x 10^9/day. The load allocation for nonpoint sources upstream from the WWTP has been set at 1.0034 x 10^{10} fcu/day, yielding a 30-day geometric mean of 100 fcu/100 mL and a reduction of almost 75 percent in nonpoint source contributions.

A combination of BMPs will be used to implement the TMDL. Public outreach and stakeholder involvement will be ongoing. New Mexico will use a long-term monitoring system that is already being used by the Surface Water Quality Bureau (SWQB). It is a rotating basin system approach to water quality monitoring. A select number of watersheds are intensively monitored each year with a return frequency of 5 years. The rotating basin program will also be supplemented with other data collection efforts. There are limited available data on nonpoint sources, so additional sampling needs to be conducted to characterize upstream sources of fecal coliform bacteria. In addition to the regularly scheduled monitoring, NPDES compliance monitoring will be conducted.

Lost River, West Virginia

The Lost River is part of the Potomac River headwaters in Hardy County, West Virginia and flows northeast to the Cacapon River, then to the Potomac River and eventually to the Chesapeake Bay (USEPA Region 3, 1998). The primary land uses of the approximately 116,600-acre watershed are forest and agriculture. The designated uses of the Lost River include propagation and maintenance of fish and other aquatic life, water contact recreation, and trout water. The applicable water quality standards for the state of West Virginia are a 30-day geometric mean of 200 cfu/100 mL and an instantaneous maximum of 400 cfu/100 mL in no more than 10 percent of the samples taken in one month. The instream fecal coliform levels were occasionally above these standards in the Lost River, therefore the West Virginia Division of Environmental Protection (WVDEP) placed a 26.03-mile segment of the Lost River on the 303(d) list due to impairment by fecal coliform contamination from undetermined. EPA gathered data from various local sources (e.g., local District Conservationist, local watershed groups, national databases) to identify, characterize, and estimate potential fecal coliform loading from various land use categories distributed throughout the watershed. EPA used the Better Assessment Science Integrating Point and Nonpoint Sources (BASINS) computer model to develop the TMDL, using a hydrologically representative time period that captured the varying hydrologic and climatic conditions in the watershed.

Rio Chamita



Water Quality Indicators	TMDL	Controls
100 fcu/ 100 mL (monthly geometric mean) 200 fcu/ 100 mL maximum	WLA: 1.14x10^9 fcu/ day LA: 1.01x10^{10} fcu /day	• NPDES permit revision • Public outreach • Combination of BMPs
Source Analysis: Monitoring Link to Indicator: Divide load by flow for estimated concentration		

AR0022634

Both point and nonpoint sources were identified in the watershed. The three point sources identified for the watershed are East Hardy High School, East Hardy Early/Middle School, and the E.A. Hawse Continuous Care Center (a fifty unit nursing home). Point sources were evaluated based on available inspection reports and loads were estimated using observed average effluent flow and concentrations, where available, or permit limits for concentration and flow. The TMDL does not prescribe any load reductions from these sources since the wasteload allocation is fairly insignificant compared to the load allocation.

The watershed was broken down into seven land uses to evaluate nonpoint sources of bacteria. These seven land uses include barren, cropland, forest, other rural, pasture, residential-pervious, and residential-impervious. Failing septic systems were also identified as fecal coliform nonpoint sources to the river. Information on watershed activities were collected from published watershed studies, state and local agencies, and local watershed groups. The information was evaluated to characterize potential nonpoint sources within the watershed, by quantifying all possible sources of bacteria accumulation on the land use surface or direct input of bacteria to watershed streams. Activities contributing bacteria loads include the land application of poultry litter and cattle feedlot waste to 100 percent of cropland and 75 percent of pasture land. Failing septic systems and wildlife contributions were also identified as bacteria sources of concern. Land use sources were represented in the model with bacteria accumulation rates which were calculated based on accumulation from the various sources. For example, accumulation on pasture land was the sum of accumulation rates from the application of poultry litter and feedlot waste and from wildlife and grazing livestock.

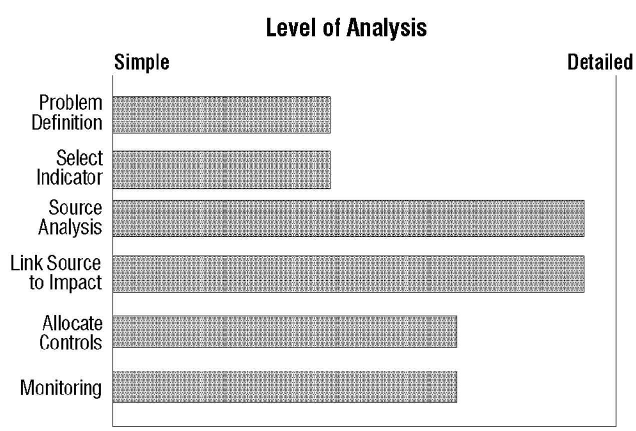
BASINS provided continuous simulation of bacteria buildup and washoff, bacteria loading and delivery, point source discharge and instream water quality response and output daily loads from each land use and point source. Existing loads were established through calibration of the model to existing water quality data. Loads were reduced until instream concentrations met water quality standards. The TMDL established necessary load reductions of 38 percent from cropland, 39 percent from pastureland, 13 percent from forest and 50 percent from failing septic systems.

Many best management practices (BMPs) were to be implemented to reduce the loading of fecal coliform bacteria to the river from nonpoint sources. Some of these BMPs include composting, increased transport of litter to less vulnerable areas, and vegetated buffer strips to prevent delivery of fecal coliform to the Lost River. Periodic monitoring of fecal coliform bacteria in a number of locations throughout the Lost River watershed has been conducted for many years and was scheduled to continue.

Chickasawatchee Creek, Georgia

USEPA Region 4 completed a fecal coliform TMDL for each of 42 waterbodies in the state using the same analysis methods. This summary of the Chickasawatchee TMDL (*Fecal Coliform TMDL development*

Lost River



Water Quality Indicators	TMDL	Controls
200 cfu/ 100 mL 30-day geometric mean 400 cfu/ 100mL instantaneous maximum in no more than 10% of samples taken in a month	• No load reductions from point sources • 38% reduction in cropland • 39% reduction in pasutreland • 13% reduction in forest • 50% reduction in septics	• Composting • Transport of litter • Vegetated buffer strips
Source Analysis: BASINS model Link to Indicator: BASINS model		

AR0022635

Chickasawhatchee Creek Watershed, undated) provides an example of one of the 42 TMDLs. The Chickasawatchee Creek watershed is located in Terrell, Calhoun, and Dougherty counties, Georgia in the Flint River Basin. The creek's designated use is fishing. The surrounding land uses include urban-pervious, urban-impervious, agriculture/pastureland, forest, and barren. Chickasawatchee Creek was placed on Georgia's 303(d) list due to more than 20% of water samples having a fecal coliform concentration of greater than 400 cfu/100 mL. The water quality standards in Georgia are different for summer and winter. From May-October the standards are a 30-day geometric mean of 200 cfu/100 mL. From November-April the standards are a geometric mean of 1,000 cfu/100 mL with an instantaneous maximum of 4,000 cfu/100 mL.
There is one permitted wastewater treatment facility in the watershed, Dawson WPCP. The standard monthly average effluent limitation contained in Georgia's NPDES permits is 200 cfu/ 100 mL. Potential nonpoint sources of fecal coliform bacteria include baseflow and the five different land uses in the watershed. Baseflow includes septic tank seepage, leaking sanitary sewer pipes, illicit sewer connections, and animal feedlots.

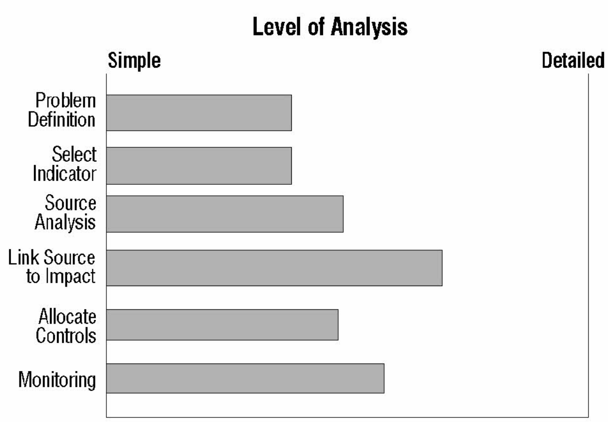
USEPA's BASINS model was used to derive the linkages between the measured fecal coliform levels in the stream and the sources of fecal coliform. The parameters needed to run the model were derived or estimated from existing land use data, rainfall data, available stream geometry information, land slope data, soil characteristics, literature values, and best professional judgement. There are many activities and land uses that contribute to the fecal coliform loading to the stream system at various rates and time. Therefore, many allocation scenarios for the TMDL were developed to reflect different reduction strategies for the various sources and their respective loadings. One of the allocation scenarios that achieves the target value of 175 cfu/100 mL is shown in Table 2-5.

Table 2-5. Load reductions to Chickasawatchee Creek

Land Use	Percent Load Reduction
Baseflow	94%
Agriculture/Pastureland	25%
Urban-impervious	50%
Urban-pervious	50%
Forest	0%
Barren	50%

The model indicates that the fecal coliform loading from agricultural runoff, urban runoff, and baseflow are the primary sources of impairment to the stream. This TMDL is based on the limited amount of readily available fecal coliform data that was used to put the stream segment on the Georgia 303(d) list. No watershed or stream-specific modeling data were collected, therefore, this TMDL is primarily useful for making screening level decisions, useful as one factor to priority rank the watersheds for additional monitoring or for planning the implementation of pollution controls. Additional monitoring of the stream was recommended to increase the confidence of the model results. If additional modeling shows continued exceedance of the water quality standards, more data would be collected to develop a better model.

Chickasawatchee Creek



Water Quality Indicators	TMDL	Controls
<u>May to Oct.</u> 200 cfu/ 100 mL (30-day geometric mean) <u>Nov. to April</u> 1,000 cfu/ 100 mL (30-day geometric mean) <u>Instantaneous Maximum</u> 4,000 cfu/ 100 mL	Various allocations possible	
Source Analysis: BASINS model Link to Indicator: BASINS model		

Problem Identification

Objective: Identify background information and establish a strategy for specific 303(d) listed waters that will guide the overall TMDL development process. Summarize the pathogen-related impairment(s), geographic setting and scale, pollutant sources of concern, and other information needed to guide the overall TMDL development process and provide a preliminary assessment of the complexity of the TMDL (what approaches are justified and where resources should be focused).

Procedure: Inventory and collect data and information needed to develop the TMDL. Information collected should include an identification of degree and type of water quality standards impairment and preliminary identification of sources, numeric targets, proposed analytical methods, data needs, resources required, and possible management and control techniques. Interview watershed stakeholders and local, state, tribal, and federal agency staff to identify all information relevant to the waterbody and its watershed. Establish plans for incorporating public involvement in the development of the TMDL. Revise the problem definition as new information is obtained during TMDL development.

Key Questions to Consider for Problem Identification

1. What are the designated uses and associated impairments?
2. What data are readily available?
3. What is the geographic setting of the TMDL?
4. What temporal considerations affect the TMDL?
5. What characteristics of the waterbody and/or its watershed might be exacerbating or mitigating the problem?
6. What are the sources of the pollutant and what are the pathways it might take to reach the waterbody?
7. How will margin of safety and uncertainty issues be addressed in the TMDL?
8. What are some potential control options?

OVERVIEW

Developing a TMDL requires formulating a strategy that addresses the potential causes of the water quality impairment and available management options. The characterization of the causes and pollutant sources should be an extension of the process originally used to place the waterbody on the 303(d) list. Typically, the impairment that caused the listing will relate to water quality standards being exceeded—either pollutant concentrations that exceed numeric criteria or waterbody conditions that do not achieve a narrative water quality standard or support the designated use. In many cases, the problem is self-evident and its identification will be relatively straightforward. In other cases, the complexity of the system might make it more difficult to definitively state the relationship between the pathogen sources and the impairment.

The following key questions should be addressed during the initial strategy-forming stage. Answering these questions results in defining the approach for developing the TMDL. A problem statement based on this problem identification analysis is an important part of the TMDL because it relates the TMDL to the 303(d) listing and describes the context of the TMDL, thereby making the TMDL more understandable and useful for implementation planning.

KEY QUESTIONS TO CONSIDER FOR PROBLEM IDENTIFICATION

1. What are the designated uses and associated impairments?

The goal of developing and implementing a TMDL is to attain and maintain water quality standards in an impaired waterbody to support designated uses. With that in mind, TMDL developers should stay focused on addressing the pathogen-related problem interfering with the designated uses. The problem identification should answer the following:

- How are water quality criteria expressed (narrative or numeric criteria, average or instantaneous concentration)?
- What nonattainment of standards caused the listing?
- What data or qualitative analyses were used to support this decision?
- Where in the waterbody are designated uses supported and where are they impaired?

- What are the critical conditions, in terms of flow and season of the year, during which designated uses are not supported?
- How do pathogens affect the designated uses of concern (e.g., Do the presence of pathogens in the water at a bathing beach create a health hazard?)

States, tribes, and other jurisdictions commonly compare measurements of various physical, chemical, and biological indicators to established water quality standards to determine whether waters support designated or existing uses such as recreation, fish and shellfish harvesting and consumption, and domestic drinking water supply. Exceedances of water quality standards evident through comparison of existing monitoring data to water quality criteria usually form the basis for listing the waterbody.

For human pathogens, routine monitoring data for specific viruses, fecal indicator bacteria, or protozoans might be collected for the waterbody. Routine monitoring is usually conducted for sources of drinking water and shellfish harvesting, and at recreational beaches. Densities of total coliform bacteria and fecal coliform bacteria are frequently measured and evaluated. One of the most important issues for pathogen loading assessments is that the presence of bacterial indicators does not always prove or disprove the presence of human pathogenic bacteria, viruses, or protozoans.

Documented nonsupport of the designated use may cause a waterbody not to attain water quality standards, whether in combination with exceedance of numeric criteria or without any criteria exceedances. Public complaints of disease associated with use of the surface waters could be a factor leading to listing of the waterbody. Epidemiologic data, including reports of diseases that might be caused by waterborne pathogens, are collected by the Centers for Disease Control and Prevention (and published in *Morbidity and Mortality Weekly Report*), and surveys are conducted for diseases that occur following exposure to contaminated water (e.g., acute gastroenteritis, hepatitis, cholera, ear infections). Existing epidemiologic data might be used to help identify waterbodies that might pose disease risks to humans, in particular diseases caused by microorganisms that are not or cannot be identified by routine monitoring methods. Both the CDC and EPA acknowledge that waterborne outbreak and disease data are vastly under reported.

Problem Identification in Maquoit Bay

The Maquoit Bay watershed in Maine covers an area of 7,878 acres and primarily comprises three land uses—forest (60%), agriculture (13%), and residential (12%). The remaining 15% is divided among roads, wetlands, and a small amount of commercial land. Fecal coliform bacteria have been identified as a potential source of contamination to the bay, affecting both water quality and the economically important shellfish resource. Shellfish closures due to high fecal indicator concentrations have been problematic for years. Water quality monitoring indicates that storm water runoff from land uses in the watershed is the primary source of fecal indicators . The preliminary problem statement was:

Maquoit Bay is experiencing bacterial impairment (by fecal coliform bacteria) to water quality, resulting in the closure of nearly one-third of its productive shellfish resource. Based on an analysis of water quality data and land use practices, the primary source of the impairment was identified as runoff from the agricultural lands and failing septic systems.

To determine if a proposed zoning ordinance would result in slowing water quality impairment, additional monitoring data were collected and used to support the development of the watershed model FecaLOAD. The results of the modeling indicated that manure applications accounted for the largest load of fecal coliform, followed by failing septic systems. Actions have been taken to reduce loadings from these sources.

Source: Horsely and Witten (1996).

Recommendation: In the problem identification, identify and summarize the events leading to the listing and the data used to support the listing.

2. What data are readily available?

A waterbody is considered impaired when a water quality standard is violated, whether through exceedance of a numeric or narrative criterion, impairment of designated use, or violation of an antidegradation policy. It is important that the data and rationale used to list the waterbody as impaired be made available to staff responsible for developing the TMDL. In addition to water quality monitoring data, documentation for the listing of waters based on narrative standards or other information should also be provided.

As much as possible, managers should identify the problem based on currently available information, including water quality monitoring data, watershed analyses, best professional judgement, information from the public, and any previous studies of the waterbody (e.g., state and federal agency reports, university sponsored studies, reports prepared by environmental organizations). These data ideally will provide insight into the nature of the impairment, potential pathogen and indicator bacteria sources, and pathways by which pathogens and indicator bacteria enter the waterbody.

Managers should also compile data that will be needed for actual development of the TMDL during the problem identification stage. These data likely will include the following:

- Water quality measurements (e.g., enterococci concentrations).
- Waterbody size and shape information (e.g., volume, area, depth, width, length).
- Waterbody flow and runoff information.
- Tributary location and contributions (flow and water quality).
- Watershed land uses and land use issues.
- Meteorological data (temperature and precipitation).
- Soil surveys and geologic information.
- Topographical information.
- Information on local contacts.
- Past studies/surveys, which may include source water assessments conducted under the SDWA.

Maps of the watershed also will be invaluable, either hard copies, such as USGS quad maps, or (if available) electronic files or GIS systems. Point sources, known nonpoint sources, and land uses should be identified on these maps to provide an overview of the watershed and to identify priority areas for pathogen and/or indicator bacteria loading caused by human activities.

Information on related assessment and planning efforts in the study area should also be collected. TMDL development should be coordinated with similar efforts to reduce TMDL analysis costs, to increase stakeholder participation and support, and to improve the outlook for timely implementation of needed control or restoration activities. Examples of related efforts that should be identified include:

- State, local, or landowner-developed watershed management plans.
- Source water protection activities under the SDWA.
- Nonpoint source control projects.
- Stormwater management plans and permits.
- Natural Resource Conservation Service (NRCS) conservation plans, Environmental Quality Incentives Program (EQUIP) projects, and Public Law 566 (PL-566) small watershed plans.
- Land management agency assessment or land use plans (e.g., Federal Ecosystem Management Team [FEMAT] watershed analyses or Bureau of Land Management [BLM] proper functioning condition assessments).
- Clean Lakes program projects.
- Comprehensive monitoring efforts (e.g., National Water Quality Assessment [NAWQA] and Environmental Monitoring and Assessment Program [EMAP] projects).

Recommendation: Contact agency staff responsible for the waterbody listing and collect any information they have available. Contact other relevant agencies, including state natural resources, water resources, fish and wildlife, and public health agencies, and state drinking water and source water protection administrators and prepare an inventory of available information. Universities are often a good source of data for a waterbody.

3. What is the geographic setting of the TMDL?

TMDLs can be developed to address various geographic scales. The geographic scale of the TMDL primarily will be a function of the impairment that prompted the waterbody listing, the type of waterbody impaired, the spatial distribution of use impairments, and the scale of similar assessment and planning efforts already under way.

The selection of TMDL scale may involve trade-offs between comprehensiveness in addressing all designated use and source issues of concern and the precision of the analysis. Table 3-1 summarizes the advantages and disadvantages of developing TMDLs for larger (i.e., greater than 50 mi^2) and smaller (less than 50 mi^2) watersheds.

Recommendation: When the designated use impairments are at the bottom of a watershed (e.g., in a lake or estuary), address the entire watershed at once by using less-intensive, screening-level assessment methods. Follow-up monitoring can assess the

Table 3-1. Advantages and disadvantages of different TMDL watershed analysis scales

	Large TMDL Study Units (> 50 square miles)	Small TMDL Study Units (< 50 square miles)
Advantages	• Accounts for watershed processes operating at larger scales • More likely to account for cumulative effects • Avoids need to complete separate studies for multiple tributaries	• Easier to identify and address fine-scale source-impact relationships and to identify needed control actions • Possible to use more accurate, data-intensive methods
Disadvantages	• Confounding variables obscure cause-effect relationships • Numeric target setting harder for heterogeneous waterbody features • Source estimation more difficult because land areas more heterogeneous • Lag time between pollutant discharge and instream effects potentially longer, effectiveness of source controls therefore harder to detect • Analysis may not give sufficient detail to provide allocations at the scale of the waterbody listing	• May miss cause-effect relationships detectable only at broad scale (cumulative impacts) • May necessitate many separate TMDL studies in a basin

effectiveness of the pathogen or indicator bacteria reduction and, if necessary, more in-depth analysis can target specific high-priority areas within the watershed that have local problems.

When impairments occur throughout a watershed, the analysis should be conducted for smaller, more homogenous analytical units (i.e., subwatersheds). For example, specific river reaches that are impaired might require detailed TMDLs to address upstream point and nonpoint sources. If this subwatershed approach is chosen, care should be taken to apply consistent methodologies from one subwatershed to the next so that an additive approach eventually can apply to the larger watershed.

4. What temporal considerations affect the TMDL?

TMDLs must consider temporal (e.g. seasonal or interannual) variations in discharge rates, receiving water flows, and designated use impacts. These considerations are especially important for stream pathogen TMDLs because both point and nonpoint pathogen sources can discharge at different rates during different time periods, causing the critical conditions for a pathogen TMDL to vary.

For example, point sources or continuous loading sources (e.g., wastewater treatment plants) tend to have the greatest impact on stream water quality under low-flow, dry weather conditions, when dilution is minimal. The lowest in-stream flows normally occur in summer or early fall when in-stream temperatures are high.

Nonpoint loading sources that may deliver bacteria loads (e.g., surface runoff from pasture) are typically precipitation-driven. Storm event producing surface runoff can wash-off bacteria deposited and accumulated on the land surfaces, resulting in the delivery of sometimes significant loads of bacteria to the receiving waterbody. Maximum impacts from rain-related nonpoint source loading generally occur at high flows.

The critical conditions of impairment are determined by the source behavior. Often, sources of bacteria are diverse and occur in combination. For example, a stream may receive bacteria loads from such direct sources as watering livestock and illicit sewer connections and from runoff from agricultural areas. Varying sources can result in multiple critical conditions. In some cases, it may be necessary to evaluate a TMDL under a variety of conditions to account for the different times of greatest impact from sources (e.g., low flow and high flow). Analysts may want to identify the different critical conditions and evaluate them separately. Another option is to develop the TMDL for a time period that encompasses all of the possible critical conditions. For example, develop a TMDL based on various flow rates or develop separate TMDL allocations for different seasons. When using dynamic modeling, a representative time period can be chosen for the TMDL development to represent conditions likely to occur (i.e., a year with wet and dry seasons, a multiple-year period to account for meteorologic and source variations).

Seasonal variations are also important for pathogen TMDLs. In-stream concentrations of bacterial indicators and pathogens vary over the course of the year in response to many factors, including weather and source characteristics. For example, the coliform removal efficiency may be lower during winter months, resulting in less die-off than in warmer months. Source behavior may also influence the seasonal variability of bacterial loading in a watershed. Significant bacterial loads can originate from agricultural land receiving land application of manure; however, farmers may only spread manure during the spring season, resulting in high spring loads and lower summer, fall and winter loads.

Several states have bacterial indicator standards that vary based on season. These standards usually correspond with the seasonal use designation (e.g., primary contact recreation for summer months and secondary recreation for the non-summer months). TMDLs are developed for waters exceeding the applicable water quality standards and are applied according to the conditions of those standards (e.g., criteria set for the season or flow). Therefore, if a waterbody exceeds a seasonal designated or existing use or criterion, a TMDL is developed and applied on a corresponding seasonal basis.

Recommendation*:* Address temporal considerations during the problem identification stage of TMDL development to ensure that a good strategy is in place as the specific technical components of the TMDL are completed. Specific guidance on addressing temporal issues is provided in each section of this protocol.

5. What characteristics of the waterbody and/or its watershed might be exacerbating or mitigating the problem?

The problem identification is based on an evaluation of available data to gauge whether water quality conditions and loadings are causing impairment of the waterbody. If information concerning likely future stresses to be placed on the watershed (e.g., development projects, industrial use proposals) is available, it should also be included. Waterbodies currently impaired by pathogens, as well as good-quality waters, can be significantly affected by alterations in land use. For example, pathogen loading might increase if incorrectly designed, sited, operated, or maintained septic systems are built, if more cows are grazed in a pasture adjacent to a stream, if a marina is added to a lake, or if wildlife populations increase in a protected forest. Pathogen loading can decrease if sewage treatment plants are upgraded, manure application to cropland is properly managed, or discharges from boats are prohibited. Evaluation of monitoring data over one or more years, as well as evaluation of all available information on the resources, trends, and policies potentially affecting pathogen loading in the watershed, is needed to develop the most effective and appropriate TMDL for the watershed. The data should be reviewed to develop an understanding of the spatial (throughout the watershed) and temporal (e.g., seasonal, daily) variation in densities of pathogens. Data from special analyses for specific pathogens in water and in fish and shellfish and epidemiologic surveys of diseases in humans and animals that come into contact with or ingest the surface water could help identify the major health concerns.

Recommendation*:* Identify any characteristics of the watershed and waterbody and predictions of future use that might affect the TMDL analysis.

6. What are the sources of the pollutant and what are the pathways it might take to reach the waterbody?

During the problem identification, the TMDL developer should first understand the relative magnitude of the various indicator bacteria and/or pathogen sources, including identifying when loading occurs and how pathogens enter the waterbody. Any practice that might result in human or animal fecal matter entering a waterbody, including runoff from the land surface, direct discharges, and contaminated groundwater flowing into surface waters, should be considered as a potential source of human pathogens. Using readily available information, it is possible to identify potential point sources of pathogen loading and provide a preliminary determination of land uses in the watershed, as well as potential "hot spots" for nonpoint sources of pathogens (e.g., runoff from pastures and feedlots, wildlife). Land uses provide important clues to the sources of pathogens in the watershed (e.g., forest, pastureland, concentrated animal operations, impervious surfaces in urban areas).

In addition, information on ineffective treatment, failures, or bypasses under high flow conditions from wastewater treatment plant discharges should be included. The problem statement should include relevant information on the characteristics of the waterbody and its watershed, especially characteristics or conditions that might exacerbate or mitigate the problem (e.g., size of the watershed, land uses, topography, soil data, climatological data, reservoir depth, residence time). Any other complicating factors that could potentially contribute to the problem should also be included.

Regardless of the pollutant or source, the TMDL should demonstrate an understanding of the entire process of pollutant delivery and impact, including the role that bacteria play in affecting the impairment, even when the stressor is self-evident. In other cases, care must be taken to ensure that the correct relationship between the pollutant and the impairment is identified. An understanding of the physical process of pathogen loading should include all potential sources of pathogens (including runoff from nonpoint sources, rainfall-driven point sources, and WWTP discharges), the transport of pathogens, and mixing processes in a waterbody that affect pathogens; the biological relationship between pathogen survival and light availability, temperature, salinity, and pH; and the chemical process(es) by which the pathogen density might increase or decrease (e.g., processes that influence the availability of nutrients and organic compounds in the water column and sediments).

It is often helpful to prepare a schematic that illustrates, in words, diagrams, or pictures, how different ecosystem processes interact with the pollutants and their sources to cause the waterbody impairment. Some of these processes could substantially alter pathogen loading or affect human health concerns. For example, the viability of bacteria, viruses, and protozoans could be greatly reduced in a shallow stream with low turbidity that receives high levels of ultraviolet radiation compared to a very turbid stream. Water currents at the mouth of an estuary might significantly dilute the load, or they could concentrate the pathogens in a protected cove. Consideration of the effects of environmental factors and processes affects how the TMDL allocations are determined.

Recommendation: Conduct an inventory of available information on point sources using information available from state or local agencies or databases such as the EPA's Permit Compliance System (PCS). For nonpoint sources, identify all possible land use-specific sources through analysis of aerial photos, land cover maps or databases, and information from federal, state, and local agencies. When using maps or GIS coverages to determine land uses, document the scale, resolution, and date of the information. In large watersheds, the only available data might be at a small scale and the ability to conduct field verification will be limited. In smaller watersheds, the utility of the same data might be limited because the scale and minimum mapping unit might hide important details, but field verification of such data is possible. In all cases, rely on the best and most relevant data sets, document all issues related to scale and date, and verify analysis with field visits.

Prepare a flowchart or schematic detailing the processes that might affect impairment of the waterbody (see Figure 3-1). In the schematic, identify the critical pathways and processes of the pollutant and the relative magnitude of the sources. The schematic will help provide a visual guide to what information is still needed to conduct the analysis.

7. How will margin of safety and uncertainty issues be addressed in the TMDL?

Considerable uncertainty is usually inherent in estimating pathogen loading from nonpoint sources, as well as predicting water quality response. The effectiveness of management measures (e.g., support of agricultural BMPs) in reducing loading is also subject to significant uncertainty. These uncertainties, however, should not delay development of the TMDL and

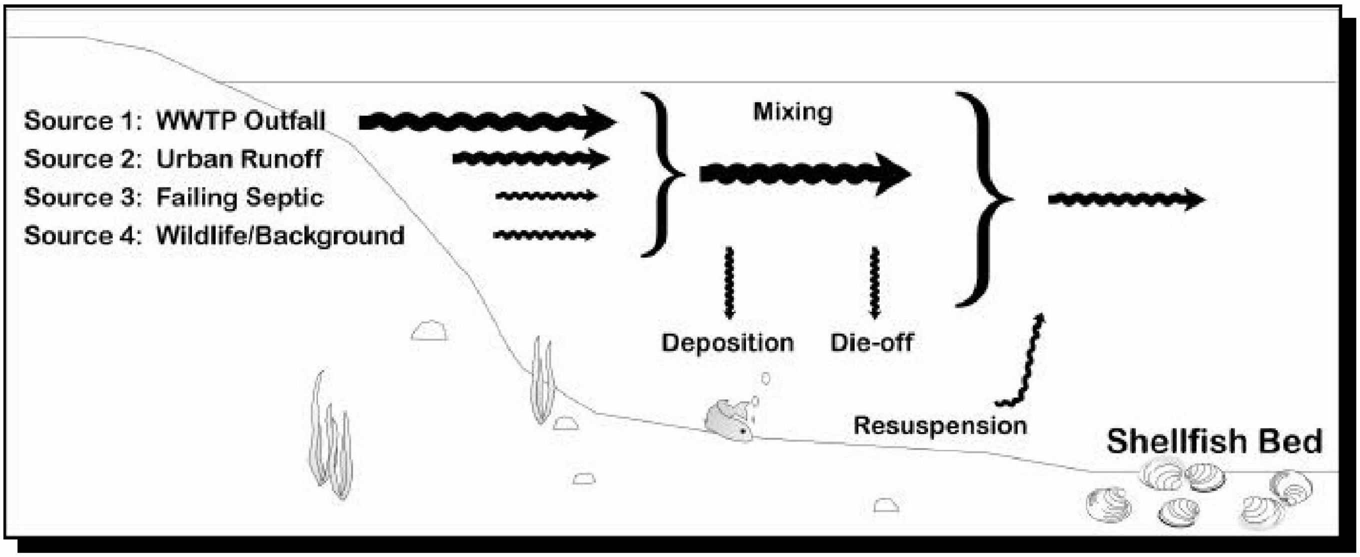


Figure 3-1. Example schematic showing processes important to waterbody impairment

implementation of control measures. EPA regulations (40 CFR 130.2(g)) state that load allocations for nonpoint sources "are best estimates of the loading which may range from reasonably accurate estimates to gross allotments, depending on the availability of data and appropriate techniques for predicting the loading." USEPA (1991a; 1999) advocated the use of a phased approach to TMDL development as a means of addressing these uncertainties. Under the phased approach, load allocations and wasteload allocations are calculated using the best available data and information, recognizing the need for additional monitoring data to determine if the load reductions required by the TMDL lead to attainment of water quality standards. The approach provides for the implementation of the TMDL while additional data are collected to reduce uncertainty.

When using models during the development of the TMDL, either to predict loadings or to simulate water quality, managers should address the inherent uncertainty in the predictions. Various techniques for doing so include sensitivity analysis, first-order analysis, and Monte Carlo analysis. These techniques are briefly summarized in Section 6 and are also discussed in various documents (e.g., IAEA, 1989; Cox and Baybutt, 1981; Chapra, 1997; Reckhow and Chapra, 1983).

TMDLs also address uncertainty issues by incorporating a margin of safety into the analysis. The margin of safety is a required component of a TMDL and accounts for the uncertainty about the relationship between pollutant loads and the quality of the receiving waterbody (CWA section 303(d)(1)(c)). The results of the uncertainty analysis performed for any modeling predictions can be factored into the decision regarding a margin of safety. The margin of safety is traditionally either implicitly accounted for by choosing conservative assumptions about loading and/or water quality response, or is explicitly accounted for during the allocation of loads. For example, a margin of safety is explicitly set at 5×10^7 cfu/day (or 10 percent of the loading capacity of 5×10^8 cfu/day) with the remainder of 4.5×10^8 cfu/day allocated as wasteload and load allocations. Table 3-2 lists several approaches for incorporating margins of safety into pathogen TMDLs.

Table 3-2. Approaches for incorporating margins of safety into pathogen TMDLs

Type of MOS	Available Approaches
Explicit	• Do not allocate a portion of available pathogen loading capacity; reserve for MOS
Implicit	• Conservative assumptions in pathogen loading and transport rates • Conservative assumptions in the estimate of pathogen control effectiveness • Conservative assumptions in deriving the numeric target (e.g., set lower than water quality criteria)

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Recommendation: During the problem identification process, the TMDL developer should decide, to the extent possible, how to incorporate a margin of safety into the analysis. The degree of uncertainty associated with the source estimates and water quality response should be considered with the value of the resource and the anticipated cost of controls. In general, greater margins of safety should be included when there is more uncertainty in the information used to develop the TMDL. It may also prove feasible to include margins of safety in more than one TMDL analytical step. For example, relatively conservative numeric targets and source estimates could be developed that, in combination, create an overall margin of safety adequate to account for uncertainty in the entire analysis.

8. What are some potential control options?

The problem identification should begin to identify potential management alternatives, such as BMPs and load reduction from point sources. A general level of understanding should be reached concerning the relative load reductions that must be obtained from point versus nonpoint sources and whether uncontrollable pathogen sources are a significant factor. If no obvious level of pathogen/indicator bacteria control will achieve the designated use of the waterbody, the appropriateness of the applicable water quality standard should be evaluated.

The problem statement should identify and stress the opportunity to take advantage of any ongoing watershed protection efforts. It should also address coordination with other state agencies (e.g., human health and pollution control agencies) and federal agencies (e.g., USEPA, U.S. Department of Agriculture, U.S. Geological Survey, U.S. Department of Health and Human Services, U.S. Forest Service, Bureau of Land Management, Department of the Interior) to avoid duplication of effort. In some cases, related watershed studies (e.g., CWA section 319, Clean Lakes, USDA PL-566) might provide the basis for many elements of the TMDL.

Local organizations can also be instrumental in developing grassroots protection programs for waterbodies, and they should be included in the problem statement formulation. For example, the town of Orleans, Massachusetts, developed and installed several remediation options to reduce fecal indicator loading and protect shellfish harvesting waters from nonpoint sources (Bingham et al., 1996).

Recommendation: Identify and document all ongoing efforts, including watershed characterization efforts, restoration efforts, and volunteer monitoring activities by local stakeholders. Include all efforts, regardless of the scale. Many local watershed groups support volunteer monitoring programs for specific stream reaches that may be a very small segment of the impaired waterbody.

RECOMMENDATIONS FOR PROBLEM IDENTIFICATION

- Identify events resulting in the 303(d) listing and the data to support the listing. Include any data or anecdotal information that supports qualitative approaches to develop the TMDL.
- Identify the specific role pathogens play in affecting designated or existing uses, usually through qualitative judgment and consultation with experts.
- Contact agency staff responsible for the waterbody listing and collect any available information.
- Prepare a flowchart or schematic detailing the processes that might affect waterbody impairment.
- Conduct an inventory of available information on point or nonpoint sources using information available from state or local agencies or databases.
- Identify temporal (e.g., seasonal) factors affecting such issues as discharge rates, receiving water flows, and designated use impacts. Temporal considerations will affect all subsequent stages of TMDL development for pathogens.
- Identify and document all current watershed restoration or volunteer monitoring efforts.
- Identify any characteristics or future uses of the watershed or waterbody that might affect the TMDL analysis.

RECOMMENDED READING

(Note that a full list of references is included at the end of this document.)

USEPA. Undated. *TMDL Case Study Series*. <http://www.epa.gov/OWOW/tmdl/case.html>. U.S. Environmental Protection Agency, Washington, DC.

USEPA. 1991a. *Guidance for Water Quality-based Decisions: The TMDL Process.* EPA 440/4-91-001. U.S. Environmental Protection Agency, Assessment and Watershed Protection Division, Washington, DC.

USEPA. 1995a. *Watershed Protection: A Project Focus*. EPA 841-R-95-003. U.S. Environmental Protection Agency, Office of Water, Washington, DC.

USEPA, 1995b. *Watershed Protection: A Statewide Approach*. EPA 841-R-95-001. U.S. Environmental Protection Agency, Office of Water, Washington, DC.

USEPA. 1996a. *TMDL Development Cost Estimates: Case Studies of 14 TMDLs*. EPA R-96-001. U.S. Environmental Protection Agency, Office of Water, Washington, DC.

USEPA 1999. *Draft Guidance for Water Quality-based Decisions: The TMDL Process.* 2nd ed. EPA 841-D-99-001. U.S. Environmental Protection Agency, Washington, DC. <http://www.epa.gov/owow/tmdl/proprule.html>.

Identification of Water Quality Indicators and Target Values

Objective: Identify numeric or measurable indicators and target values that can be used to evaluate the TMDL and the restoration of water quality in the listed waterbody.

Procedure: Select one or more indicators that are appropriate to the waterbody and local conditions. Key factors to consider include scientific and technical validity, as well as practical issues such as cost and available data. Identify target values for the indicator(s) that represent achievement of water quality standards and are linked (through acceptable technical analysis) to the reason for waterbody listing.

OVERVIEW

To develop a TMDL, it is necessary to have a quantitative measure that can be used to evaluate the relationship between pollutant sources and their impact on water quality; such measurable parameters are called *indicators* in this document. For pathogen TMDLs, indicators will often be based on state water quality standards developed to protect human health from exposure to pathogens in surface waters. The standards establish designated uses for a waterbody and the narrative or numeric water quality criteria necessary to support those uses, as well as the development of an antidegradation policy. The standards developed for pathogen pollutants are usually based on the detection of generic groups of microorganisms that have been associated with fecal contamination and that indicate pathogenic microorganisms are likely to be present in the water. These standards are the basis on which a waterbody's impairment by pathogens is determined. Pathogen impairments may be identified through either the violation of a numeric water quality standard (e.g.,criterion for *E. coli* or enterococcus bacteria) or the nonattainment of a waterbody's designated or existing use (e.g., primary contact recreation).

Key Questions to Consider for the Identification of Water Quality Indicators and Target Values

1. What is the water quality standard that applies to the waterbody?
2. What factors affect indicator selection?
3. What water quality measures could be used as indicators?
4. What are appropriate target values for the chosen indicators?

Water quality standards consist of the following elements:

- Designated uses
- Numeric and narrative criteria for supporting each use
- Antidegradation statement (40 CFR Part 131)

This section of the protocol provides background on water quality standards and their relationship to TMDL indicators, lists various factors that should be addressed in choosing a TMDL indicator, and provides recommendations for setting target values under different circumstances.

KEY QUESTIONS TO CONSIDER IN THE IDENTIFICATION OF WATER QUALITY INDICATORS AND TARGET VALUES

For many TMDLs, the numeric target will be determined directly by the numeric criteria associated with the state water quality standards. However, in those cases where the numeric criteria are not available or are not protective of designated or existing uses, the use of an alternative or supplementary fecal indicator may be required (Figure 4-1). Whether using the water quality standards or numeric targets for other indicators, a number of factors should be considered.

1. What is the water quality standard that applies to the waterbody?

Federal recommended microbiological water quality criteria have been developed to provide guidance to states for the establishment of their own standards for identifying pollution problems. These standards are often used as the TMDL target value. These criteria, which are based on indicator organisms, are summarized in Table 4-1. The microbiological indicators yield a general assessment of water quality and safety for the designated or existing use and do not identify specific human pathogens; that is, the exceedance of criteria developed

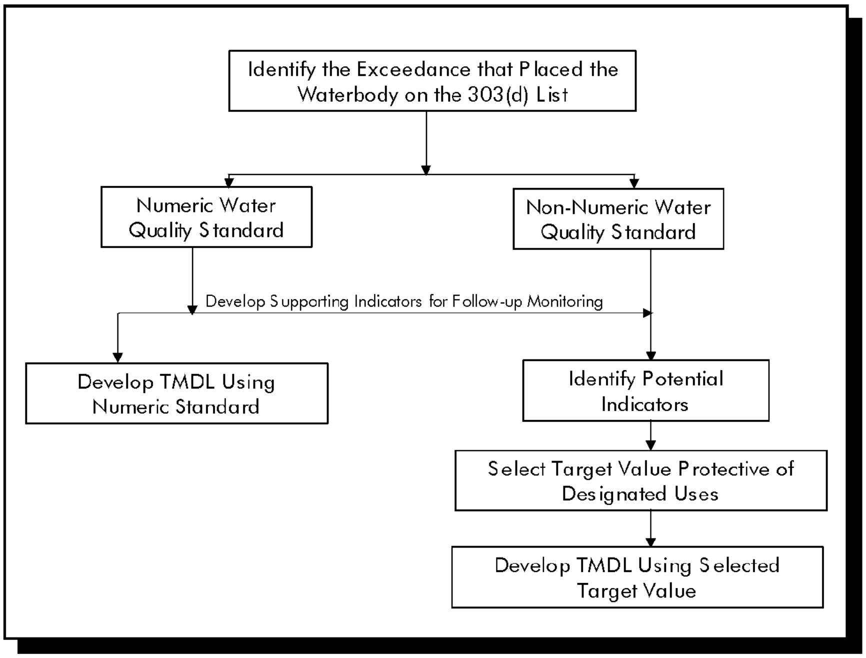

Figure 4-1. Factors for determining indicators and endpoints

for *E. coli* and enterococci bacteria indicates that the water *might* cause some type of illness following exposure to that water. For example, recreational use by swimmers or surfers could be impaired by the presence of high densities of fecal indicator bacteria because some of those fecal indicator bacteria might cause gastrointestinal illness if the water is swallowed; commercial or recreational harvesting of oysters in an estuary could be impaired because the presence of high densities of these fecal indicators suggests that other human pathogens such as the infectious hepatitis A virus might have accumulated in the shellfish tissues.

Some jurisdictions have adopted the federal 304(a) criteria for enterococci or *E. coli*. (See box defining 304(a) criteria). EPA publishes 304(a) criteria as guidance to states and tribes. States and tribes may adopt EPA's 304(a) criteria, 304(a) criteria modified to reflect site-specific conditions or criteria based on other scientifically defensible methods. A 1998 summary of state standards used in the United States for recreational waters can be found at http://www.epa.gov/ost/beaches/local/statept.pdf; however, the most recent standards should be obtained from the particular jurisdiction because changes might have occurred since the publication of this document.

It should be noted that several states have fecal indicator standards that vary based on season, usually in conjunction with seasonal use designations (e.g., primary contact recreation for summer months and secondary recreation for the rest of the year). TMDLs are developed for waters violating the applicable water quality standards and are applied according to the conditions of those standards (e.g., criteria set for a season or flow). Therefore, if a waterbody violates a seasonal designated or existing use or criterion, a TMDL is developed and applied on a corresponding seasonal basis.

Where a numeric water quality standard exists and is an appropriate indicator for use attainment, the TMDL analysis should use the standard. In some cases, the waterbody of concern has a numeric water quality standard that might not appropriately or sufficiently reflect the use impairment, and the use of a supplementary indicator or set of indicators might provide additional means for measuring attainment of designated or existing uses. For example, if a waterbody meets its established numeric criteria for *E. coli* or enterococci but does not support its designated use of primary contact recreation, a TMDL must be developed for the waterbody.

The term "water quality criteria" is used in two sections of the Clean Water Act—section 304(a)(1) and section 303(c)(2). Section 304(a)(1) requires the Administrator of EPA to publish criteria for water quality accurately reflecting the latest scientific knowledge on the kind and extent of all identifiable effects on health and welfare that may be expected from the presence of a pollutant in any body of water. Under section 303, water quality criteria associated with specific stream uses form the basis for enforceable water-quality based limits in CWA permits when adopted as state water quality standards. It is not until their adoption as part of the state water quality standards that the criteria become regulatory. The water quality criteria adopted in the state water quality standards could have the same numerical limits as the criteria developed under section 304. However, in many situations, states may want to adjust the water quality criteria developed under section 304 to reflect local environmental conditions and human exposure patterns before incorporating those criteria into the state water quality standards.

Table 4-1. Currently recommended criteria for indicators of elevated levels of pathogens

Designated Use	Pathogens Evaluated	Criteria
Recreation Primary (e.g., swimming, surfing, diving) Secondary (e.g., wading, boating)	*E. coli* [a]	Freshwater: Geometric mean of 126 CFU per 100 mL, based on not less than 5 samples equally spaced over a 30-day period; no sample should exceed a one-sided confidence limit (CL) calculated using the following as guidance: designated bathing beach - 75% CL; moderate use for bathing - 82% CL; light use for bathing - 90% CL; infrequent use for bathing - 95% CL; based on a site-specific log standard deviation, or if site data are insufficient to establish a log standard deviation, then using 0.4 as the log standard deviation
	Enterococci [a]	Freshwater: Geometric mean of 33 CFU per 100 mL, based on not less than 5 samples equally spaced over a 30-day period; no sample should exceed a one-sided confidence limit (CL) calculated using the following as guidance: designated bathing beach - 75% CL; moderate use for bathing - 82% CL; light use for bathing - 90% CL; infrequent use for bathing - 95% CL; based on a site-specific log standard deviation, or if site data are insufficient to establish a log standard deviation, then using 0.4 as the log standard deviation Marine: Geometric mean of 35 CFU per 100 mL, based on not less than 5 samples equally spaced over a 30-day period; no sample should exceed a one-sided confidence limit (CL) calculated using the following as guidance: designated bathing beach - 75% CL; moderate use for bathing - 82% CL; light use for bathing - 90% CL; infrequent use for bathing - 95% CL; based on a site-specific log standard deviation, or if site data are insufficient to establish a log standard deviation, then using 0.7 as the log standard deviation
	Fecal coliform [b]	Geometric mean of 200 CFU per 100 mL, based on not less than 5 samples equally spaced over a 30-day period and no more than 10 percent of the samples exceeding 400 CFU per 100 mL during any 30-day period. *[Note: fecal coliform criteria are used by many states; however, EPA recommends the use of the E. coli and enterococci criteria.]*
Shellfish harvesting waters	Total coliform [b]	Geometric mean of 70 MPN per 100 mL, with not more than 10 percent of the samples taken during any 30-day period exceeding 230 MPN per 100 mL.
	Fecal coliform [b]	Median concentration should not exceed 14 MPN per 100 mL with not more than 10 percent of the samples taken during any 30-day period exceeding 43 MPN per 100 mL.
Public drinking water sources	Total coliform [c]	Ninety percent of daily raw water samples $\leq$ 100 CFU/100 mL for surface water systems to remain unfiltered
	Fecal coliform [c]	Ninety percent of daily raw water samples $\leq$ 20 CFU/100 mL for surface water systems to remain unfiltered
	E. coli [d, e]	Lakes and Reservoirs - 10 CFU/100 mL as annual average Flowing Streams and Rivers - 50 CFU/100 mL as annual average
	Cryptosporidium oocysts [d, f]	0.075 oocysts/L (7.5 oocysts/100 L) to avoid upgrading treatment

[a] Source: Federal 304(a) Ambient Water Quality Criteria for Bacteria (USEPA, 1986)
[b] Source: Quality Criteria for Water (USEPA, 1976)
[c] See 40 CFR 141.71(a)(1) and sampling frequency table under §141.74(b)(1)
[d] These provisions are scheduled to be proposed in the Spring of 2001
[e] For a small system (<10,000) that tests for *E. coli* as a surrogate for *Cryptosporidium*, exceeding an *E. coli* threshold would require that system to either test directly for *Cryptosporidium* or to upgrade its treatment.
[f] The current treatment requirement for all surface water systems is 2 logs (99%) removal. Sampling results >0.075 oocysts/L would trigger a requirement to upgrade to 3 logs (99.9%) removal or inactivation; >1 oocyst/L would trigger a requirement to provide 4 logs (99.99%) removal or inactivation; and >3 oocysts/L would trigger a requirement to provide 4.5 logs (99.995%) removal or inactivation of *Cryptosporidium*.

It would be helpful to use a supplementary indicator that is more clearly linked to the designated use impairment (e.g., *Cryptosporidium* when cases of cryptosporidiosis are associated with use of the waterbody) and establish a corresponding target value to develop a TMDL.

Designated uses are the desirable uses that the water quality should support. Examples are drinking water supply, primary contact recreation (e.g., swimming), and aquatic life support. Each designated use has a unique set of water quality requirements or criteria that must be met for the use to be realized. Waterbodies may be designated for multiple uses (USEPA, 1995c).

Recommendation: Determine the water quality standard for the waterbody. Use the water quality standard when it is numeric and it represents the best available measure of designated or existing use impairment. Use supplementary indicators when the numeric standard is not an appropriate measure of designated or existing use support. When a numeric standard is used, note any important issues, including where the standard is applied (e.g., end of pipe or open water, segment or entire length), number of samples required, averaging period, applicable time period (e.g., summer months) and number of exceedances allowed.

2. What factors affect indicator selection?

Even when attainment of designated or existing uses can be measured using numeric water quality standards, other factors should be considered before developing the TMDL. The factors include the relative value of the waterbody, staff expertise available for monitoring and analyzing data for other indicators, and resources available. For example, the primary drinking water source for a large population might rely on a different fecal indicator than that used for secondary contact recreation waters. In other cases, the designated use might be impaired despite no observed violation of the numeric criteria. For these situations, alternative or supplementary fecal indicators might need to be evaluated. In addition, sampling protocols might need to be modified to better examine the concentration of the fecal indicators.

Most pathogen-impaired waterbodies are of concern because of the human health risks associated with exposure to the pathogens. Table 4-2 presents some of the pathogens associated with sewage that can cause disease following exposure. Where information is available on the concentrations of these pathogens in the environment, their infectivity, and sources, they all qualify for use as possible indicators. However, for most of these pathogens, this type of information is lacking and pathogens are often difficult to reliably detect using simple and inexpensive laboratory methods.

Table 4-2. Human pathogens likely to be associated with sewage

Bacteria	Viruses	Protozoa
Aeromonas hydrophila	Adenovirus	*Entamoeba histolytica*
Bacillus anthacis	Coxsackie A and B	*Acanthamoeba* spp.
Campylobacter pylori	Echovirus	*Giardia* spp.
Campylobacter spp.	Hepatitis A	*Cryptosporidium*
Clostridium botulinum	Non-A, non-B hepatitis	
Clostridium perfringens	Norwalk/Snow Mountain/ small round viruses-related gastroenteric viruses	
Escherichia coli	Parvovirus	
Helicobacter	Poliovirus	
Klebsiella penumoniae	Reovirus	
Listeria monocytogenes	Rotavirus	
Mycobacterium spp.		
Pseudomonas spp.		
Salmonella spp.		
Shigella spp.		
Staphylococcus aureus		
Vibrio spp.		
Yersina spp.		

Sources: Ahmed, 1991; Kennish, 1992; McNeill, 1992; Koenraad et al., 1997

Some criteria that should be considered during the selection of an indicator are the following: it should be easily detected using simple laboratory tests, it should not be present in unpolluted waters, and it should appear in concentrations that can be correlated with the extent of contamination (Thomann and Mueller, 1987). Table 4-3 presents potentially useful indicators, including coliform and enterococcus bacteria. These indicators satisfy many of the criteria suggested by Thomann and Mueller (1987) and are used in many state water quality standards.

Recommendation: Select an appropriate fecal indicator based on the information known about and the impairment to the waterbody. Consider the established water quality standard, alternative indicators, designated or existing use, and resources. Document all steps in the process, and, if possible, involve stakeholders in the decisions. If a pathogen is

Table 4-3. Some potential indicator organisms for TMDL development

Group	Indicator Organisms[a]
Viruses	F1 coliphage; MS2 bacteriophage; poliovirus type 1 strain Lsc2ab; enteroviruses
Coliform bacteria	**Total coliform**; **fecal coliform**; ***Escherichia coli***; *Klebsiella* spp.
Enterococcal Bacteria	*Streptococcus faecalis; Streptococcus faecium*
Protozoa	*Cryptosporidium* spp. *Giardia* spp.

[a] Water quality standards often exist for the indicator organisms in bold type.

considered for use as an appropriate fecal indicator, consult a trained sanitary/environmental microbiologist.

3. What water quality measures could be used as indicators?

As discussed earlier in this section, EPA publishes 304(a) criteria as guidance to the states and tribes in establishing their water quality standards. Current 304(a) criteria recommendations as related to pathogens are a geometric mean of 126 CFU/100 mL for *E. coli*, a freshwater geometric mean of 33 CFU/100 mL for enterococci, and a marine geometric mean of 35 CFU/100 mL for enterococci (USEPA, 1986). EPA believes *E. coli* and enterococci are more accurate indicators of the presence of pathogens than fecal coliform bacteria. Therefore, the current 304(a) criteria suggest the use of *E. coli* or entorococci bacteria, replacing the 1968 criteria (USEPA, 1968), which recommended a geometric mean of 200 CFU/100 mL for fecal coliform.

Presently, most states are using the 1968 water quality criteria for fecal coliform bacteria in their water quality standards. However, since 1986, the EPA has recommended the use of *E. coli* and enterococci bacteria as indicators of pathogenic contamination in waterbodies.

States and tribes may adopt EPA's 304(a) criteria, 304(a) criteria modified to reflect site specific conditions, or criteria based on other scientifically defensible methods. The state must develop the TMDL using the current, approved, state water quality standards. If states or tribes do not have their own bacterial water quality criteria, then the federal criteria should be used. If the state chooses at any time to revise its criteria, then the standards need to be revised and approved according to state procedures, which typically include public notification and review and approval by the EPA Standards Branch at the Regional EPA office. The Alaska decision (*Alaska Clean Water Alliance* v. *Clark* (1997)) has set the precedent in regard to states revising their water quality standards. The rule states that standards submitted to EPA after the effective date of the rule do not become "applicable" water quality standards for CWA purposes until approved by EPA, and that "applicable" standards remain the CWA standards until EPA approves state or tribal revisions or publishes replacement water quality standards (USEPA, 2000).

States may also use additional indicators (i.e., alternative bacteria, protozoa, viruses) for tracking and analysis purposes as long as the TMDL is written to meet the current applicable water quality standards. For example, the water quality standards may not always reflect the actual problem, such as a cryptosporidiosis outbreak. A protozoan, such as *Cryptosporidium*, can be used as an indicator in cases where *Cryptosporidium* is known to be the pathogen of concern in the waterbody. If this is the case, the TMDL needs to show that once the target for *Cryptosporidium* is met, the water quality standards will also attained.

Recommendation: Many states and tribes presently use the 1968 fecal coliform water quality criteria as indicator values. The EPA, however, recommends the use of *E. coli* and enterococci as bacterial indicators, as stated in the federal 304(a) criteria. Regardless of what indicator states use to develop the TMDLs, either from state water quality standards or alternate indicators, the TMDL must be written to result in the attainment of water quality standards. Therefore, the states must establish a relationship between the indicator used and existing state water quality standards to prove that meeting the indicator target value will correlate to attainment of water quality standards.

4. What are appropriate target values for the chosen indicators?

For the indicators used in developing pathogen TMDLs, a desired or target condition must be established to provide measurable environmental management goals and a clear linkage to attaining the applicable water quality standards. In the case of pathogen TMDLs, the target values for most indicators are already established directly through the numerical criteria in state water quality standards. These water quality criteria can be used or a more stringent or more appropriate value can be used as the target value.

Often, states have multiple parts to their standards. For example, standards may express a "not to exceed," instantaneous criteria as well as a geometric mean based on a minimum number of samples collected in a specific time frame. The availability of data and nature of the impairment may dictate which part of the standard should be used as the target. For example, the geometric mean criteria used for the bacterial indicator target value may be based on at least 5 samples collected in a 30-day period. As many monitoring programs are based on quarterly sampling, there may not be enough historical data to support the use of the geometric mean criteria as the target. In this case the "not to exceed" value may be used. For example, the recommended federal criteria for enterococci at freshwater bathing beaches is 33 CFU/100 mL, based on not less than 5 samples equally spaced over a 30-day period. Unfortunately, data containing 5 samples taken at equal intervals throughout the month are often not available. In this case, the "not to exceed" criteria of a one-sided confidence limit of 75% for enterococci bacteria should not be exceeded according to the federal criteria for freshwater bathing beaches.

Before developing the TMDL, it is necessary to determine the appropriate target value. In most cases the state water quality criteria will be the appropriate target. If the state standards contain multipart criteria, it should be decided whether it is necessary to use one or all parts as the target.

If the state water quality criteria do not reflect the impairment or problem, alternate indicators should be used and appropriate target values established. The target value must be set at a level that represents the attainment of the current water quality standards.

Recommendation: The target values for most bacteria indicators are already established directly through the numeric criteria in state water quality standards. The TMDL must be written to attain these standards. If an alternate indicator is used, the TMDL must establish some relationship between the water quality standards and the alternate indicator, showing that the target value represents attainment of water quality standards.

RECOMMENDATIONS FOR IDENTIFICATION OF WATER QUALITY INDICATORS AND TARGET VALUES

- When appropriate, use the established water quality standard as the numeric target for TMDL development.
- Select a fecal indicator based on its scientific and technical appropriateness and information known about the waterbody, including the established water quality standard, the identified impairment, supplementary indicators, designated or existing use, and resources, while considering practicality and cost. Document all steps in the process and involve stakeholders in the decisions.

RECOMMENDED READING

(Note that a full list of references is included at the end of this document.)

Francy, D.S., D.N. Myers, and K.D. Metzker. 1993. Escherichia coli *and fecal-coliform bacteria as indicators of recreational water quality. Water Resources Investig. Rep.* 93-4083, U.S. Geological Survey, Earth Science Information Center, Denver, CO.

USEPA. 1994a. Guidelines for deriving site-specific water quality criteria for the protection of aquatic life and its uses. Chapter 4 in *Water Quality Standards Handbook*. U.S. Environmental Protection Agency, Office of Water Regulations and Standards, Washington, DC.

USEPA. 1986. *Ambient water quality criteria for bacteria–1986.* EPA-A440/5-84-002. U.S. Environmental Protection Agency, Washington, DC.

USEPA. 1998. *Bacterial water quality standards status report.* EPA 823/R-98/003. U.S. Environmental Protection Agency, Office of Water, Washington, DC.

Source Assessment

Objective: Characterize the type, magnitude, and location of sources of fecal indicator loading to the waterbody.

Procedure: Compile an inventory of all possible sources of pathogens to the waterbody. Sources may be identified through assessment of maps, data, reports, and/or field surveys. It is likely that a combination of techniques will be needed depending on the complexity of the source loading and watershed delivery processes. After compiling an inventory, monitoring, statistical analysis, modeling, or a combination of methods should be used to determine the relative magnitude of source loadings.

OVERVIEW

The source assessment is needed to evaluate the type, magnitude, timing, and location of loading to an impaired waterbody. It further describes the sources initially identified during the problem identification. Several factors should be considered in conducting the source assessment. These factors include the various types of sources (e.g., point, nonpoint, background), the relative location and magnitude of loads from the sources, the transport mechanisms of concern (e.g., runoff, direct deposit), and the time scale of loading to the waterbody (duration and frequency of fecal indicator loading to receiving waters).

Once sources have been identified and a relative ranking of their contribution has been conducted, the loadings from each source should be estimated using a variety of techniques, including relying on existing monitoring data, doing simple calculations, performing spreadsheet analysis using empirical methods, or using one or more of a range of computer modeling systems. The selection of the appropriate technique is an outgrowth of the problem identification and watershed characterization performed during the initial phase of TMDL development.

Key Questions to Consider for the Source Assessment

1. What are the potential sources of pathogens to the waterbody of concern?
2. How can sources of pathogens to the waterbody of concern be characterized?
3. How should sources be grouped for assessment and load allocation?
4. How can pathogen loads be estimated?

A TMDL should include an evaluation of all the sources contributing to the fecal indicator loading of the waterbody. The detail of the assessment will vary, however, depending on the overall approach best suited to the site-specific conditions. The selection of the appropriate method for estimating loads should be based on the complexity of the problem, the time constraints, the availability of resources and monitoring data, and the management objectives under consideration. It is usually advantageous to select the simplest method that addresses the questions at hand, uses existing monitoring information, and considers the available resources and time constraints for completing the TMDL. This section of the protocol describes various types of sources, identifies procedures for characterizing loadings, and introduces a process for selecting a source assessment technique.

KEY QUESTIONS TO CONSIDER FOR SOURCE ASSESSMENT

1. What are the potential sources of pathogens to the waterbody of concern?

Pathogens are delivered to waterbodies by a wide variety of point and nonpoint sources (see box on page 5-2). Treated municipal sewage is a point source of bacterial, viral, and protozoal contamination. Not all human pathogens are removed or rendered harmless by treatment processes. Periodic effluent overflows and high-flow bypass from wastewater treatment plants (WWTPs) can cause occasional high loadings of pathogens. Other major point sources include combined sewer overflows (CSOs) and sanitary sewer overflows (SSOs). CSOs contribute significant pathogen loads during storm events; SSOs may contribute pathogens under both wet and dry weather conditions. Illicit discharges of residential and industrial wastes are difficult to identify but are often a major source of pathogens.

Potential Pathogen Sources

Point Sources	Nonpoint Sources
WWTPs	Domestic pets
CSOs	Animal feedlots
SSOs	Wildlife
Slaughterhouses	Septic systems
Meat processing facilities	Livestock
Poultry processing facilities	Pastures
Animal feedlots	Boat pumpout
Illicit sewage connections	Landfills
	Land application of manure
	Land application of sludge

Storm water runoff from urban watersheds might also be a significant source of pathogens, delivering pathogens present in the waste of domestic pets and wildlife and in litter. On-site wastewater systems (septic tanks, cesspools) that are poorly installed, faulty, improperly located, or are in close proximity to waterbodies are potential sources of human pathogens to surface and ground waters. Boats lacking holding tanks for pumpout also contribute potential human pathogens; marinas and waterbodies that are heavily used for recreational boating have been shown to have elevated levels of fecal indicator bacteria. Rural storm water runoff can transport significant loads of bacteria and pathogens from livestock pastures, livestock and poultry feeding facilities, and feedlots. Livestock areas with high concentrations of animal waste contribute pathogens primarily through surface runoff. Some of these sources may be subject to the requirements of the NPDES program. Facilities that process food, meat, or poultry are potential sources from overflow of holding lagoons. Manure storage and application practices might lead to pathogen loads in surface runoff depending on the time of year, the timing of the manure application with runoff-producing storm events, or the proximity of the application to the waterbody of concern. Wildlife can also contribute pathogen loadings and may be particularly important in the transmission of the protozoan pathogens *Giardia lamblia* and *Cryptosporidium*. Wildlife of concern include deer, beaver, ducks, and geese. In urban or suburban areas, large populations of deer can provide a significant source of pathogens. Although remote, pristine forested lands might appear to be unlikely candidates for pathogen sources, many wildlife species harbor microorganisms that can be pathogenic to themselves, other wildlife, and humans.

Most fecal indicators are indirect and only warn of the possibility of the presence of fecal pathogens, which are not necessarily from humans, but potentially from several sources. Current monitoring and analytical methods for coliforms and enterococci do not distinguish between indicator bacteria of human and nonhuman origin (Turner et al., 1997). Therefore, the environmental and public health implications of monitoring data are difficult to interpret in cases where contamination comes from multiple sources. This would not be a problem if there was a way to identify bacterial strains that are specific to a particular host. Other indicators and methods that permit more rapid identification of fecal indicators are under development or have been developed. Some of these alternate methods include agglutination assays, DNA hybridization tests, and polymerase chain reactions (PCR) (Koenraad et al., 1997). The PCR process shows promise for distinguishing between particular sources of fecal indicator bacteria contamination and may be used for other environmental applications. This method of microbial source tracking is known as DNA fingerprinting.

Some states have begun using DNA fingerprinting to identify sources of fecal indicator contamination in water (Pelley, 1998; Blankenship, 1996). DNA fingerprints can be used to match the genetic characteristics of bacteria in animals such as chickens, cows, and wildlife to identify pollution sources. Each animal species hosts unique strains of bacteria that are adapted to the intestinal environment of that particular host. By comparing the bacteria from the sample to fingerprints of known strains, the bacteria can be tracked to the source. DNA fingerprinting identifies the pollution source and helps managers/planners target the problem and formulate a mitigation strategy (Pelley, 1998). The current challenge is to develop a complete library of bacterial strains that is specific to each locale.

Although DNA fingerprinting has only recently been used to identify water pollution sources and is an expensive and lengthy process, it offers the promise of providing a large amount of high-quality information.

DNA Fingerprinting in Virginia

DNA fingerprinting proved helpful when a farmer on Virginia's Eastern Shore was faced with the closure of his shellfish beds due to elevated levels of *E. coli*. Failing septic tanks were assumed to be the primary source of the fecal pollution, but a survey of septic systems in the sparsely populated watershed indicated that they were not the cause and it became necessary to identify another source. The highest levels of coliform bacteria were measured in the small tidal inlets and rivulets of the wetlands located upstream of local houses, shifting suspected sources from human to other sources. Researchers collected fecal samples from raccoon, waterfowl, otter, muskrat, deer, and humans in the area and used DNA fingerprinting to confirm bacteria the suspicion that the source was not anthropogenic in nature. The DNA of the samples was analyzed and characterized, resulting in a library of more than 200 DNA patterns distributed through more than 700 *E. coli* strains. Comparing *E. coli* from the shellfish beds against the fingerprints of the known strains in the DNA library, the researchers linked the in-stream *E. coli* to deer and raccoon (mostly raccoon). Several hundred animals, including 180 raccoon, were removed from areas adjacent to the wetlands. *E. coli* levels subsequently declined by 1 to 2 orders of magnitude throughout the watershed, and previously closed or threatened areas of the tidal creeks were reopened to shellfishing.

Sources: Blankenship, 1996, and News-Notes, 1997.

Recommendation: Develop a comprehensive list of the potential pathogen sources to the waterbody of concern. Use the list of potential sources of pathogens and the watershed inventory to identify actual sources and develop a plan for identifying and accounting for the load from each.

2. How can sources of pathogens to the waterbody of concern be characterized?

Sources of pathogens can be characterized using a variety of approaches. The determination of the most appropriate techniques will be based on the extent of the problem, the size of the watershed, the availability of watershed information, the types of sources (point and/or nonpoint), and the resources available. All possible sources of information should be consulted. For example, the under the SDWA states must develop source water assessments that inventory all potential contamination sources of drinking water contaminants and their locations and state Wellhead Protection Programs typically have information on ground water recharge areas and the locations of potential contaminant sources. Polluted groundwater that is hydrologically connected to a waterbody is likely to contribute to its impairment, so potential sources of groundwater contamination should also be reviewed, particularly those near the waterbody. This information is usually available from the state drinking water or public health agency. In addition, the Safe Drinking Water Amendments of 1996 require the delineation of source water protection areas and contamination source inventories (USEPA, 1997b). Other agencies, such as USDA and state natural resources, extension service, or public health agencies, might provide useful information on the location of possible sources of pathogens.

Although agency staff can often provide significant information, other approaches can be used to identify sources, including literature and historical records searches, surveys (phone, door-to-door, windshield), and field reconnaissance, including the use of GIS data or aerial photographs (USEPA, 1991b). Reports and articles in the literature can provide useful information on past and present land uses, activities, and disturbances. Public records from which information can be obtained include registries of industrial and commercial activities, property transfer, titles, and deeds. Anecdotal information about the area should also be obtained. The local Chamber of Commerce can normally provide direction for this effort. Various survey methods can also be used to locate possible sources of contamination. Phone, mail, or personal interviews with landowners and stakeholders can often produce a significant amount of information about local sources.

Driving through the watershed is another method of identifying potential sources. This type of survey (called a windshield survey) is much less detailed than a field reconnaissance effort and can cover larger areas in less time. Field reconnaissance activities are resource-intensive and require additional resources and planning. These searches involve extensive on-site reconnaissance and may not be practical for large watersheds. Use available aerial photos from several years to identify particular sources, such as failing septic systems or land uses that generate pathogen loads (e.g., pastures). For watersheds with GIS data, good-quality land use information is often available to identify the spatial pattern of land use, as well as the proportion of coverage for each land use. Using this information, source classes can be identified to allow lumping of source information, which is especially important in watersheds with scattered nonpoint sources that are difficult to characterize independently.

For each source identified in the waterbody, note important factors such as the proximity of the source to the waterbody of concern, the processes that are important to the delivery of the pathogen to the waterbody (e.g., runoff or direct deposit), and the relative importance of each source to the overall pathogen load. Develop a pathway diagram to describe how the pollutant can enter the waterbody. Table 5-1 presents information on transport processes for potential sources of pathogens.

Recommendation: Using all available information, identify all possible sources of pathogens to the waterbody of concern. Use GIS or maps to document the location of sources and the processes important for delivery to the waterbody. Identify all government agencies and nongovernment organizations active in the watershed, and conduct interviews and collect information.

3. How should sources be grouped for assessment and load allocation?

To select appropriate analytical tools and management measures, the sources must be grouped into discrete units. The definition of each unit should be based not only on the ability of specific analytical tools to determine quantifiable loads but also on management and economic considerations. The sources should be grouped so that there is a recognizable link between sources and allocation. Although typically classified as nonpoint pollution, a groundwater contribution to pollutant loading does not fall within NPDES jurisdiction where there is a direct hydrologic connection from the facility through the groundwater to the waterbody. Therefore, such groundwater contributions to pollutant loading should be grouped with other point sources of pollutants. Grouping of sources can be accomplished by the use of database searches or matrices that identify and link these common processes or political characteristics.

By linking the common mechanisms of pollutant delivery, the appropriate analytical tools can be efficiently determined. For example, although there are different pathogen concentrations in cattle manure than in chicken manure, the delivery mechanisms are similar enough that the same analytical tool can be used to estimate the delivered load from both. An example of

Table 5-1. Sources and transport pathways for pathogens

Source/land use	Operation/activity	Samples of management activity	Frequency	Transport process(es)
Agriculture	Livestock-feedlot	Manure removal	weekly	runoff; erosion
	Livestock-manure storage	Storage structures; leachate control	variable	runoff; erosion; seepage
		Spreading schedules; storage		
	Crop-manure/sludge application	Rotation	variable	runoff; erosion
	Pasture		variable	runoff; erosion; direct
Urban/ Residential	Domestic pets	Waste pickup law	variable	runoff
	Wildlife	Management; population control	constant	runoff; direct
		Pumpout; education		
	Septic systems	Compliance	annual	leaching; interflow
	Illicit connection		constant	direct
	Landfills	Disposal	constant	runoff; leaching
Forest	Wildlife	Management; population control	constant	runoff; erosion; direct
Point Sources	WWTP	Waste treatment	constant	direct
	Slaughterhouse	Waste treatment	variable	direct
	CSOs; SSOs	Storage/transport redesign	variable	direct; rainfall-driven

Factors to Consider for Grouping Sources

- Delivery mechanisms
- Location of sources relative to waterbody of concern
- Management options under consideration
- Social, political, and economic factors
- Physical characteristics of the watershed, including slope, geology, soils, and drainage network.

social and political associations is that within the watershed there might be several different growers associations or cooperatives. It might be easier to propose management initiatives to the cooperative than to try to implement them on an individual farm basis. There might be a large source of pollutants from a major employer in the area, and reducing the loads from that source might have a significant impact on the local economy. The spatial organization of the sources is also important for identifying critical reaches to be studied. For example, a feedlot located several miles from the waterbody in the upper reaches of the watershed would usually have far less impact on loading than an equally sized parcel located next to the receiving water. Industry growth is another important factor. Chicken farm expansion can occur in a much smaller land area and at a higher density and rate than another agricultural use, such as feedlots.

Sources can also be grouped by subwatershed. For example, the watershed that is the focus of the TMDL can be divided into several smaller subwatersheds and loading estimates can be made for each of these. This approach will often be useful during the source characterization step of TMDL development, allowing for isolation of specific sources and spatial analysis of source loading and water quality response. Subwatershed delineation also makes it easier to compare the loading estimates for each subwatershed to the associated water quality observations. However, as will be discussed later in the allocation section, it will usually be necessary to group sources within the subwatersheds by land use or source categories to facilitate the allocation process.

The end result of this phase should be an efficient grouping of sources that can be evaluated using available tools and resources for the development of the TMDL. The categorization of sources may be an iterative process, depending on how well the potential groupings can be analyzed and quantifiable loads determined using the available analytical tools.

Recommendation: Group sources in a manner that establishes a link between sources and allocations, facilitates source assessment, and assists in the implementation of management actions. When grouping sources, consider location of source, pollutant transport and delivery mechanism, spatial distribution, relationship to potential control actions and necessary analytical techniques associated with the source.

4. How can pathogen loads be estimated?

The identification of sources within a watershed provides the answer to "What sources are causing the impairment?" The next step is to determine "What effects are these sources having on the waterbody?" For many sources, it is difficult to predict fecal indicator loading rates from either physical principles or national values found in the literature. Table 5-2 summarizes information from several references to illustrate the source-specific nature of fecal indicator and pathogen values. Source concentrations of pathogens and fecal indicators can also be region-specific, making site- and/or region-specific monitoring data useful, if available. Site-specific monitoring is often essential to establish accurate concentration estimates and can be combined with modeling of flow and/or sediment transport to produce load estimates. Monitoring techniques for bacterial pathogens are addressed in this protocol where they are applicable.

Estimating pathogen loads for point sources is typically easier because point sources are relatively constant in time—the discharge from a municipal wastewater treatment plant, for instance. Certain other important load sources mix the characteristics of point and episodic nonpoint sources. For example, CSOs are point sources subject to permitting, but, because they are caused by stormflow into the combined sewer system, they exhibit the episodic nature of nonpoint sources. Techniques for estimating pathogen loads to waterbodies vary according to source type and can range from qualitative assessments to detailed modeling efforts. When determining the best approach to estimate the pathogen load delivered from the source to the stream, analysts are encouraged to start with the

Table 5-2. Summary of source-specific pathogen and fecal indicator concentrations

Indicator	Concentration	Source	Reference
Clostridium perfringens	4.5×10^{7} organisms/day[a]	Duck	Roll and Fujioka, 1997
Clostridium perfringens	10^{1}-10^{3} #/mL	Raw sewage	Metcalf and Eddy, 1991
Cryptosporidium oocysts	10^{-1}-10^{1} #/mL; 0.85×10^{3} - 5.28×10^{3} #/L	Raw sewage	Metcalf and Eddy, 1991; Madore et al., 1987
Cryptosporidium oocysts	13.7 #/mL	Slaughterhouse (cattle) waste effluent	Madore et al., 1987
Cryptosporidium oocysts	1.4×10^{4} - 3.96×10^{4} #/L	Treated effluent (activated sludge only)	Madore et al., 1987
Cryptosporidium oocysts	4.0×10^{0} - 1.6×10^{1} #/L	Treated effluent (activated sludge and sand filtration)	Madore et al., 1987
Cryptosporidium oocysts	370 ± 197 oocysts/gram feces 1.2×10^{5}-3.9×10^{5} organisms/day[b]	Canada geese	Graczyk et al., 1998
Escherichia coli	2.5×10^{8} *E. coli* /day[a]	Duck	Roll and Fujioka, 1997
Escherichia coli	1.7×10^{8} *E. coli* /gram[c]	Pigeon	Oshiro and Fujioka, 1995
Enteric virus	10^{1}-10^{2} #/mL	Raw sewage	Metcalf and Eddy, 1991
Enterococci	2.0×10^{0} - 2.1×10^{5} enterococci/100 mL	Background	Overcash and Davidson, 1980
Enterococci	10^{2}-10^{3} enterococci/mL; 5.4×10^{5} enterococci/100 mL	Raw sewage	Metcalf and Eddy, 1991; Overcash and Davidson, 1980
Enterococci	2.2×10^{8} enterococci/day[a]	Duck	Roll and Fujioka, 1997
Enterococci	4.0×10^{5} enterococci/gram	Pigeon	Oshiro and Fujioka, 1995
Enterovirus	6.9×10^{0} - 2.8×10^{2} PFU/10 L	Background	Overcash and Davidson, 1980
Enterovirus	8.7×10^{2} PFU/10 L	Raw sewage	Overcash and Davidson, 1980
Fecal coliforms (FC)	1.5×10^{1} - 4.5×10^{5} MPN/100 mL	Background	Overcash and Davidson, 1980
FC	2×10^{9} organisms/day	Human	Metcalf and Eddy, 1991
FC	4.9×10^{10} organisms/day	Geese	LIRPB, 1978
FC	0.24×10^{9} organisms/day 1.4×10^{8} organisms/day	Chicken	Metcalf and Eddy, 1991 ASAE, 1998
FC	0.13×10^{9} organisms/day 9.5×10^{7} organisms/day	Turkey	Metcalf and Eddy, 1991 ASAE, 1998
FC	5.4×10^{9} organisms/day 1.0×10^{11} organisms/day 1.0×10^{11} organisms/day	Cow Cow (Dairy) Cow (Beef)	Metcalf and Eddy, 1991 ASAE, 1998 ASAE, 1998
FC	4.2×10^{8} organisms/day	Horse	ASAE, 1998
FC	11×10^{9} organisms/day 1.2×10^{8} organisms/day[a] 2.5×10^{9} organisms/day	Duck	Metcalf and Eddy, 1991 Roll and Fujioka, 1997 ASAE, 1998
FC	1.6×10^{8} organisms/gram	Pigeon	Oshiro and Fujioka, 1995

Table 5-2. Summary of source-specific pathogen and fecal indicator concentrations

Indicator	Concentration	Source	Reference
FC	8.9×10^9 organisms/day 1.1×10^{10} organisms/day	Pig	Metcalf and Eddy, 1991 ASAE, 1998
FC	18×10^9 organisms/day 1.2×10^{10} organisms/day	Sheep	Metcalf and Eddy, 1991 ASAE, 1998
FC	5×10^9 organisms/day	Dogs and cats	Horsley and Witten, 1996
FC	10^4-10^5 #/mL; 6.3×10^6 MPN/100 mL	Raw sewage	Metcalf and Eddy, 1991; Overcash and Davidson, 1980
FC	4.2×10^6 organisms/100 mL	CSO	Doran et al., 1981
FC	9.6×10^2 - 4.3×10^6 organisms/100 mL	Urban runoff	Doran et al., 1981
FC	1.2×10^2 - 1.3×10^6 organisms/100 mL	Grazed pasture runoff	Doran et al., 1981
FC	1.35×10^6 - 2.4×10^8 organisms/100 mL	Feedlot runoff	Baxter-Potter and Gilliland, 1988
FC	1.2×10^1 - 1.43×10^4 organisms/100 mL	Cropland runoff	Doran et al., 1981
Fecal streptococci (FS)	1.0×10^1 - 6.6×10^5 #/100 mL	Background	Overcash and Davidson, 1980
FS	0.45×10^9 organisms/day	Human	Metcalf and Eddy, 1991
FS	0.62×10^9 organisms/day 2.9×10^8 organisms/day	Chicken	Metcalf and Eddy, 1991 ASAE, 1998
FS	1.3×10^9 organisms/day	Turkey	Metcalf and Eddy, 1991
FS	31×10^9 organisms/day 5.9×10^{11} organisms/day 1.1×10^{11} organisms/day	Cow Cow (Dairy) Cow (Beef)	Metcalf and Eddy, 1991 ASAE, 1998 ASAE, 1998
FS	2.6×10^{11} organisms/day	Horse	ASAE, 1998
FS	8×10^9 organisms/day 8.3×10^9 organisms/day	Duck	Metcalf and Eddy, 1991 ASAE, 1998
FS	230×10^9 organisms/day 3.2×10^{11} organisms/day	Pig	Metcalf and Eddy, 1991 ASAE, 1998
FS	43×10^9 organisms/day 1.7×10^{10} organisms/day	Sheep	Metcalf and Eddy, 1991 ASAE, 1998
FS	10^3-10^4 #/mL; 1.2×10^6 #/100 mL	Raw sewage	Metcalf and Eddy, 1991; Overcash and Davidson, 1980
FS	1.7×10^6 organisms/100 mL	CSO	Doran et al., 1981
FS	1.4×10^4 - 1.7×10^6 organisms/100 mL	Urban runoff	Doran et al., 1981
FS	8.0×10^3 - 6.1×10^6 organisms/100 mL	Grazed pasture runoff	Doran et al., 1981
FS	8×10^6 - 7.9×10^7 organisms/100 mL	Feedlot runoff	Baxter-Potter and Gilliland, 1988
FS	1.7×10^3 - 3.9×10^4 organisms/100 mL	Cropland runoff	Doran et al., 1981
Giardia cysts	10^{-1}-10^2 #/mL	Raw sewage	Metcalf and Eddy, 1991

Table 5-2. Summary of source-specific pathogen and fecal indicator concentrations

Indicator	Concentration	Source	Reference
Giardia cysts	450 cysts/gram of feces 3.1x10^5 cysts/day	Canada geese	Graczyk et al., 1998
Protozoan cysts	10^1-10^3 #/mL	Raw sewage	Metcalf and Eddy, 1991
Psuedomonas aeruginosa	3.1 x 10^0 - 6.6 x 10^3 MPN/100 mL	Background	Overcash and Davidson, 1980
Pseudomonas aeroginosa	10^1-10^2 #/mL; 2.3 x 10^5 MPN/100 mL	Raw sewage	Metcalf and Eddy, 1991; Overcash and Davidson, 1980
Salmonella sp.	0 - 1.4 x 10^2 MPN/10L	Background	Overcash and Davidson, 1980
Salmonella	10^0-10^2 #/mL; 5.0 x 10^2 MPN/10 L	Raw sewage	Metcalf and Eddy, 1991; Overcash and Davidson, 1980
Staphylococcus aureus	2.5 x 10^0 - 1.2 x 10^2 MPN/100 mL	Background	Overcash and Davidson, 1980
Staphylococcus aureus	2.6 x 10^2 MPN/100 mL	Raw sewage	Overcash and Davidson, 1980
Total coliforms (TC)	10^1 - 10^6 MPN/100 mL	Background	Novotny and Olem, 1994; Overcash and Davidson, 1980
TC	7.04 x 10^{12} organisms/day 2.3 x 10^{11} organisms/day	Cow (Dairy) Cow (Beef)	ASAE, 1998 ASAE, 1998
TC	2.2 x 10^{12} organisms/day	Horse	ASAE, 1998
TC	2.7 x 10^{10} organisms/day	Pigs	ASAE, 1998
TC	5.4 x 10^9 organisms/day	Sheep	ASAE, 1998
TC	1.98 x 10^9 organisms/day	Chicken	ASAE, 1998
TC	10^5-10^9 #/mL; 10^7-10^9 MPN/100 mL; 2.3 x 10^7 MPN/100 mL	Raw sewage	Metcalf and Eddy, 1991; Novotny and Olem, 1994; Overcash and Davidson, 1980
TC	10^5-10^7 MPN/100 mL; 2.0 x 10^7 organisms/100 mL	CSO	Novotny and Olem, 1994; Doran et al., 1981
TC	10^4-10^6 MPN/100 mL	Treated effluent	Novotny and Olem, 1994
TC	5.8 x 10^4 - 2.0 x 10^7 organisms/100 mL; 10^1-10^8 MPN/100 mL	Urban runoff	Doran et al., 1981; Novotny and Olem, 1994
TC	7.0 x 10^2 - 4.9 x 10^6 organisms/100 mL	Grazed pasture runoff	Doran et al., 1981
TC	1.25 x 10^7 - 3.5 x 10^8 organisms/100 mL	Feedlot runoff	Baxter-Potter and Gilliland, 1988
TC	3.2 x 10^3 - 1.45 x 10^5 organisms/100 mL	Cropland runoff	Doran et al., 1981

[a] Converted from organisms per gram of feces using information in ASAE, 1998.
[b] Number or *Cryptosporidium* oocysts per day from geese, assuming that goose total fecal production per day is 4.5 times that of ducks (LIRPB, 1978).
[c] There is no conversion factor available to convert pigeon numbers to organisms per day.

assumption that models are not required. To select appropriate analytical tools, a number of factors should be considered, including the following:

- Availability of data and funds to support data collection
- Familiarity with the analytical tool
- Staff support
- Level of accuracy required

Depending on the complexity of the aggregate sources in the watershed, load estimation might be as simple as conducting a literature search or as complex as using a combination of long-term monitoring and modeling. The following discussion presents information on the pathogen concentrations for different sources and methods of calculating the delivery to the waterbody.

Point source loads

Loads from sewage treatment plants and industrial point sources

The greatest potential source of human pathogens is raw sewage. Raw sewage typically has a total coliform count of 10^7 to 10^9 MPN/100 mL (Novotny et al., 1989), along with significant concentrations of pathogenic bacteria, viruses, protozoans, and other parasites. Typical treatment in a municipal plant reduces the total coliform count in effluent by about 3 orders of magnitude, to the range of 10^4 to 10^6 MPN/100 mL. Most municipal plants, however, are restricted by their NPDES permit to discharge at or below the WQS for fecal coliform. Raw sewage, although usually not discharged intentionally, may reach waterbodies through CSOs, SSOs, and leaks in sanitary sewer systems.

Certain industrial processes, such as slaughterhouses and meat and poultry processing facilities, also have the potential to contribute substantial point source loads of pathogens. Analysis of loads from point sources should generally be based on the effluent monitoring required for the NPDES permit or on the permit limits, rather than use of generic assumptions, except when evaluating potential effects of proposed new sources. In analyzing such data, it should be noted that variations in concentration and load can be expected on a daily, weekly, and seasonal basis depending on the water use patterns of the community and temperature conditions within the treatment plant. Ambient upstream and downstream monitoring can also be valuable, particularly for assessing the relative contribution of point and nonpoint sources within urban settings.

In most cases, only indicator bacteria will be measured in plant effluent, rather than concentrations of specific human pathogens. Order-of-magnitude estimates of specific pathogens can, however, be obtained from information on raw sewage concentrations and effluent residual chlorine content and kill efficiency.

Loads from CSOs

One way in which raw sewage enters waterbodies directly is through CSOs. In CSOs, the pathogen load is dominated by the content of raw sewage, yet the discharge volume and degree of dilution are determined by episodic pulses of urban storm water. In many cities, combined sewer overloading by storm water and overflow events occur only a few times a year and are thus unlikely to be monitored. Typical concentrations of total coliform bacteria in CSOs are reported as 10^5 to 10^7 MPN/100 mL (Novotny et al., 1989), or about an order of magnitude greater than concentrations in treatment plant effluent. The effect of these concentrations is reduced, however, because the load is intermittent. Estimation of loading from CSOs will thus often require combining information on the pathogen or fecal indicator load of sanitary sewage with estimates of the overflow volume associated with large storm events.

Modeling the impacts of CSOs can be a difficult undertaking. Therefore, USEPA's CSO Control Policy recommends permittees take one of the following approaches: (1) *presumption approach*, consisting of meeting performance goals in minimizing the number and volume of CSO events, or (2) *demonstration approach*, requiring evidence that a CSO control plan is adequate to meet water quality standards. In either case, the CSO Control Policy expects a relatively high degree of characterization of the hydraulic operation of the sewer system in order to estimate the number and volume of CSO events. For instance, many cities simulate the hydraulic operation of the combined sewer system and storm drainage sewershed using mathematical models such as USEPA's Stormwater Management Model (SWMM). Pathogen loading is

Example: CSO Fecal Indicator Loading Assessment

A small midwestern city has a combined sewer system, subject to frequent CSO events. The combined impacts of the publicly owned treatment works (POTW) and CSOs result in exceedances of the fecal coliform criteria in the river flowing through the city. To characterize the system, the city developed a hydraulic model of its sewer system to predict overflow volumes and undertook monitoring to characterize coliform concentrations in a variety of CSO events. For initial analysis of the problem, the city calculated average FC loads during dry weather and "typical" wet weather conditions.

Under dry weather conditions, the POTW discharges on average 5,000 gal/day of effluent at an average fecal coliform concentration of 3×10^6 organisms/100 mL. During dry weather, the total load is thus

$$5000\ \frac{gal}{day} \cdot 3.785x10^3\ \frac{mL}{gal} \cdot \frac{3x10^6\ organisms}{100\ mL} = 5.7x10^{11}\ \frac{organisms}{day}$$

During an average size storm, the sewer system is able to retain and store about 40% of influent storm water. The average storm for the area is estimated to be 0.06 in/hr in intensity, with a 6.5-hr duration and about 77 hours of dry weather between storms. A storm of this size produces a runoff flow of 250 ft^3/s for this city, so the CSO discharge is (1 - 0.4) x 250 = 150 ft^3/s. Monitoring by the city indicated that the event mean concentration in CSO discharges was 1×10^7 fecal coliform organisms/100 mL. The average wet weather load is thus

$$\frac{1x10^7\ organisms}{100\ mL} \cdot 150\ \frac{ft^3}{s} \cdot 28.317x10^3\ \frac{mL}{ft^3} \cdot 3600\ \frac{s}{hr} \cdot 6.5\frac{hr}{day} = 9.9x10^{15}\ \frac{organisms}{day}$$

The long-term average, made up of both wet and dry periods, may be estimated as

$$\frac{(77hr \cdot 5.7x10^{11}\ \frac{organisms}{day}) + (6.5\ hr \cdot 9.9x10^{15}\ \frac{organisms}{day})}{77\ hr + 6.5\ hr} = 7.7x10^{14}\ \frac{organisms}{day}$$

then typically estimated by measuring the typical distribution of pathogen concentrations in sanitary sewage, calculating the concentration resulting from dilution of sewage by stormflow during a storm event, and simulating the discharge of overflows at this concentration in response to the rainfall event. Detailed information on modeling and monitoring for CSOs is provided in WPCF (1989) and Nix (1990).

Nonpoint source loads

Nonpoint sources of pathogen loads are typically separated into urban and rural categories since runoff and load generation processes differ systematically between these environments. In urban or suburban settings with high amounts of paved impervious area, important sources of loading are the washoff of contaminated refuse in surface stormflow and leakage of sanitary sewer systems. In rural settings, the amount of impervious area is usually much lower and important sources of pathogen load may include diffuse runoff of animal wastes associated with the erosion of sediments, runoff from concentrated animal operations, and failing or illicitly connected septic tanks.

Most nonpoint loads result from stormwater washoff, and load estimation requires both flow volume and pollutant concentration in runoff. Relatively simple modeling techniques can provide good estimates of surface stormflow volume, in both urban and rural settings. Modeling of the pathogen concentration in stormflow is considerably more difficult, however, and generally results in a calibration exercise against measured in-stream data. The data available for use in calibration often limit the accuracy ultimately achievable in simulation models of nonpoint pathogen loads. Modeling is typically conducted for single endpoints such as fecal coliform bacteria. There is currently little or no experience modeling other pathogens such as *Cryptosporidium, Giardia*, and viruses. In assessing nonpoint loads of fecal indicator bacteria, both the size of the load and the timing of loading are of interest. Both urban and rural nonpoint sources can generate large fecal indicator loads during individual washoff events. Therefore, analysis of loading from precipitation-driven nonpoint sources must consider both "how much" and "how often." The example in the box below presents a scoping-level analysis of "how much" and "how often" for runoff from an animal feedlot.

The fecal indicator load in runoff from animal operations is highly variable and is affected by the runoff characteristics of the feeding area, number and condition of animals, and extent of implementation of BMPs for runoff control. There is no substitute for site-specific monitoring and local experience, but in many cases initial estimates must be made without site-specific monitoring.

Urban storm water loads[1]

Estimating urban storm water loads is complicated by a lack of data and high variability in available monitoring data. Both viruses and pathogenic bacteria have been detected in storm runoff from urban areas at densities high enough to suggest a potential health risk. Indeed, coliform concentrations in urban stormwater can be of the same order of magnitude as concentrations in treatment plant effluent. The origins of urban fecal indicator loads are diverse and can include leakage from sanitary sewers and direct loading of human fecal matter, as well as fecal indicator bacteria derived from dog and cat feces.

Pathogen loads in urban storm water can be estimated using techniques at a variety of levels of complexity, ranging from very simple techniques using loading rate assumptions and constant concentration estimates, to statistical estimates, to buildup/washoff simulation.

[1] Some urban stormwater is considered a point source by the CWA, and is subject to NPDES permitting. However, the techniques used to assess urban stormwater are more characteristic of nonpoint sources, and so are described in this section.

Watershed-scale models suitable for TMDL development are summarized in USEPA (1997b).

The FecaLOAD model is an example of a simple technique that uses hydrogeological and meteorological factors such as soil properties related to the suitability for sewage disposal, distance of source from surface water, and precipitation and runoff relationships (Horsley and Witten, 1996) to qualitatively rank potential bacteria sources and distribute them in the model. The model was developed for input that is relatively easy to obtain or estimate (e.g., from the county soil survey). The model then uses the inputs to calculate outputs, by land use volume of runoff, loading of fecal coliform, and average concentration of fecal coliform in runoff. The FecaLOAD model was developed and applied for the evaluation of bacterial loading to Maquoit Bay in Brunswick and Freeport, Maine (Horsley & Witten, 1996)

Constant concentration estimates assume that all runoff has the same concentration. Note that this simple approach is often combined with sophisticated flow modeling of storm water, in which case the result might give an accurate picture of load timing even though time variability in concentration is not simulated. An obvious question is what constant concentration to use. One option is to use values reported in the literature. Literature values can be highly variable, however, and it is preferable to use site-specific measurements because of large site-to-site variability. Another option is to obtain values from the information provided in NPDES permits. Some urban stormwater dischargers are subject to a non-continuous discharge NPDES permit. The permit is based on the frequency of discharge, the total mass of discharge, the maximum rate of discharge of pollutants, and the prohibition or limitation of specified pollutants by mass, concentration, or other measure.

Statistical or regression approaches provide a little more sophistication by attempting to relate expected concentration to characteristics of the watershed. For instance, Glenne (1984) proposed a simple regression relationship between total coliform concentration in surface runoff and population density in the watershed. Regression approaches are developed based on site-specific relationships and have limited transferability. Finally, buildup and washoff of pollutants on urban impervious surfaces can be simulated directly. This method is the ostensibly physically based approach incorporated into many popular storm water models, such as SWMM and the Hydrological Simulation Program-Fortran (HSPF). Buildup refers to all of the complex spectrum of dry-weather processes that deposit or remove pollutants between storms, including deposition and street cleaning. These processes lead to accumulation of material associated with solids that is then "washed off" during storm events. Models incorporating buildup and washoff functions can also account for pathogen die-off with a component for pollutant decay transformation.

Failing or illicitly connected septic systems

Septic systems can potentially contribute significant pathogen loads to receiving waterbodies due to system failure and surface or subsurface malfunctions. In some cases, local health departments can provide information on failing septic systems (e.g., location, frequency, failure rates). However, in many watersheds, the specific incidences and locations of malfunctioning systems is unknown, which makes the task of characterizing the impact of pathogen loads from failing septic systems difficult. There are, however, methods of estimating the distribution of failing septic systems in a watershed using available information on the occurrence of failing systems or failure rates in a particular area, or where no site-specific information is available, county statistical data and literature values. For example, the National Small Flows Clearinghouse (NSFC) surveyed approximately 3,500 local and state public health agencies about the status of onsite systems across the country (NSFC, 1993) and provides the number of reported failing septic systems in the U.S. by county. Using the county-specific estimates from NSFC (1993), the number of failing septic systems in a county can be extrapolated to the watershed level based on county and watershed land use distribution. The number of failing systems also can be estimated by applying some appropriate failure rate, either from literature or professional judgment, to the total number of septic systems in a watershed. Local agencies or data from the U.S. Census Bureau can provide estimates of total septic systems in a state or county. County-level population, demographic and housing information, including septic tank use, can be retrieved from the U.S. Census Bureau by choosing the appropriated state and county on <www.census.gov/datamap/www/> or by searching the Summary Tape File 3A database on the U.S. Census Bureau website.

In addition to distribution or number, characteristics about the discharge of failing systems is necessary to evaluate their contribution of pathogen loads. If site-specific information on system effluent is not available, literature values are available on the typical concentrations of septic system effluent (Horsley & Witten, 1996) and typical effluent discharge rates (Metcalf and Eddy, Inc., 1991). Table 5-2 provides some values that might be useful in characterizing effluent from a failing septic systems. Because information on bacteria indicator and pathogen concentrations in septic effluent is limited, it may be appropriate to use available literature values for raw sewage or untreated effluent. Although using site-specific information is typically preferred, in cases where data are not available, the use of concentrations typical of raw sewage results in the incorporation of an implicit margin of safety into the analysis through the likely overestimation of the actual indicator bacteria concentration.

Rural nonpoint loads

The rural nonpoint sources of pathogen load of greatest concern are typically associated with animal operations, in which large quantities of fecal matter are generated. Pathogens from these areas can reach waterbodies through direct runoff or after the waste has been spread on fields. For instance, improper application of manure to frozen land surfaces can result in periodically high loads of pathogens and nutrients. Land application of municipal waste biosolids can also be a significant source of pathogen load. Regardless of the presence of obvious sources, such as land application of biosolids, a background loading rate resulting from the net inputs of domestic animals, wildlife, and leaking septic systems can always be expected.

As with urban loads, rural nonpoint loads may be estimated using techniques at a variety of levels of complexity, ranging from loading function estimates to use of complex simulation models. The loading function approach simply assigns an estimated average rate of pathogen loading to a given land use. Such an approach is appropriate for scoping long-term average loads, typically on an annual basis, but it cannot capture

Example: Characterizing Fecal Coliform Loads from Failing Septic Systems

The watershed of Buck Creek in Baker County was subdivided into 10 subwatersheds. Literature values, land use information and Census Bureau data were used to estimate the number of failing septic systems in each of the 10 watersheds and their contributing fecal coliform load. NSFC (1993) reported 305 failing septic systems in Baker County. Without knowing the spatial distribution of septic systems, functioning or failing, it was assumed that failing systems are distributed evenly throughout the county. Using the total area of the county (658,093 acres), the density of failing septic systems for the county was calculated as follows

$$\frac{305 \text{ failing systems}}{658{,}093 \text{ acres}} = 0.00046 \, \frac{\text{failing systems}}{\text{acre}}$$

The county density of failing systems was then multiplied by the area of each subwatershed in the county to estimate the number of failing systems in each subwatershed. For example, the Buck Creek 1 subwatershed within the Buck Creek watershed is 8,520 acres in size and is contained completely within Baker County. The estimated number of failing septic systems in Buck Creek 1 is

$$0.00046 \, \frac{\text{failing systems}}{\text{acre}} \times 8{,}520 \text{ acres} = 4 \text{ failing septic systems}$$

Literature values and Census Bureau data were used to estimate the loading from the failing septic systems in Buck Creek 1 using a representative effluent flow and concentration. Horsley & Witten (1996) estimates septic effluent concentrations as 10^6 counts/100 mL with an average daily discharge of 70 gallons/person/day. U.S. Census Bureau county data was used to estimate the average number of people per household that might be served by septic systems. Using this information, the load from failing septic systems within the Buck Creek 1 subwatershed is estimated as follows:

$$4 \text{ failing systems} \times \frac{10^6 \text{ counts}}{100 \text{ mL}} \times \frac{70 \text{ gal}}{\text{person} \cdot \text{day}} \times 2.6 \, \frac{\text{person}}{\text{household}} \times 3785.2 \, \frac{\text{mL}}{\text{gal}} = 2.76 \text{ x } 10^{10} \, \frac{\text{counts}}{\text{day}}$$

This is a simplified example that does not take into account the die-off or attenuation of loadings of fecal coliform from failing septic systems to the stream. This assumption of the worst case scenario can be used in developing the margin of safety for the TMDL.

the intermittent nature of precipitation-driven loads. Further, one of the most important determinants of rural nonpoint load is the extent of adoption and efficiency of best management practices (BMPs), such as use of animal waste storage and detention ponds, riparian buffer zones, and proper timing and methodology for field application of wastes. Site-specific analysis rather than use of generic loading functions is usually appropriate.

More sophisticated approaches are based on the simulation of surface runoff and movement of sediment and solids. Indicator bacteria and pathogen loading is incorporated into such models by assumptions regarding the concentration present in solids and a pollutant delivery ratio. At one extreme, estimates of surface runoff may be combined directly with representative runoff concentrations to provide a rough estimate of the time series of loading (McElroy et al., 1976). At the other extreme are detailed models of rainfall, runoff, and erosion processes accounting for variability in both space and time, such as AGNPS (Young et al., 1986). Like urban storm water models, rural storm water models are capable of providing a reasonable representation of flow and, to a lesser degree, sediment transport. Accurate estimation of indicator bacteria or pathogen load beyond the scoping level, however, will be almost entirely dependent on site-specific calibration.

Various agricultural activities can be significant sources of rural nonpoint source pathogen loads within a watershed. Livestock produce manure that may contain pathogens as well as fecal bacteria. The manure and the associated pathogens may be deposited directly in watershed streams or on land surfaces where it may be transported to streams through stormwater runoff. Information helpful in identifying, characterizing, and quantifying agricultural sources of pathogens include:

- Livestock counts or densities (#/acre for pastures and feedlots).
- Livestock confinement and grazing schedules.
- Access of livestock to watershed streams.
- Application schedules and rates for agricultural waste (e.g., poultry litter, cow manure).
- Locations of feedlots (if unavailable in land use coverage).
- Manure production estimates and waste characteristics.

Local agencies (e.g, extension offices, NRCS, Soil Water and Conservation Districts) are often an excellent source of information on agricultural activities and management practices within the watershed. In addition to local agencies, good sources of data concerning agricultural activities are watershed studies, university studies, and USDA's Census of Agriculture [http://www.nass.usda.gov/census/census97/highlights/ag-state.htm]. Among other data, the Census of Agriculture provides state and county information on distribution of agricultural land uses, farm sizes, and livestock inventories and sales.

The consideration of the distribution of livestock, both temporally and spatially, is important when evaluating the contribution of bacteria loads from agricultural activities. At any time, cattle in a watershed may be confined, grazing in pastures, or watering in stream reaches. Where and when cattle or other livestock are contributing bacteria loads determines the behavior, transport and impact of the load. Cattle in pastures deposit pathogen loads on the land surface where they accumulate and are available for washoff and transport to receiving waterbodies. The number of livestock in pastures and the amount of time spent grazing should be considered in the evaluation of livestock as a source of bacteria loads.

Cattle within pastures may have access to watershed streams and spend time watering. Unlike livestock depositing manure (and fecal bacteria) on pasturelands, livestock watering in stream reaches can contribute significant loads of bacteria directly to stream reaches. Local agencies should be consulted for characterizing the potential loads from various agricultural activities within the watershed. Information on fencing and grazing practices in the watershed, the percentage of cattle with access to streams or grazing pastures within proximity to streams, whether site-specific or assumed on judgment, is useful in estimating the direct contribution of manure and bacteria from livestock to streams. Estimated time spent in streams is also key in the analysis. For example, direct contributions will likely be higher in summer months when cattle spend more time cooling in watershed streams.

When cattle are confined, the manure produced might be collected and spread on pastures and cropland. The application of cattle manure (and other agricultural waste such as poultry litter) can provide a significant source of fecal bacteria to land surfaces. Many states or localities have guidelines on manure spreading practices (e.g., timing, amount). Information on application rates and schedules can assist in appropriately representing the contribution of bacteria from the land application of agricultural waste in the TMDL analysis. Also, some of these operations are regulated by NPDES permits, and the reporting for these permits may contain information useful in calculating a load rate.

Groundwater-surface water interactions

Pathogens are of concern in both surface and ground water, and can move between the two media. Under USEPA's Enhanced Surface Water Treatment Rule, groundwater sources of drinking water that are under the direct influence of surface water are generally treated the same as surface water sources because the water table is so close to the surface there is no appreciable attenuation of pathogen loading from the surface. Contaminants discharged to ground water can also affect surface water. For instance, septic fields near streams can load pathogens to a stream through ground water transport, particularly very small diameter pathogens such as viruses. In most geologic settings, however, such routes are of minor significance compared to the potential load from malfunctioning (surface-discharging) septic systems. One major exception is karst limestone areas, in which surface and ground water flow may be freely interconnected by solution cavities.

Example: Estimation of Fecal Coliform Loads from Grazing Beef Cows in a Watershed

Laurel River watershed contains areas in Gunston and Putnam counties. A major potential source of bacteria loading within the watershed is grazing livestock, primarily cattle. Data from the 1997 USDA Census of Agriculture provided numbers of livestock in each county covering portions of the watersheds, as well as total pastureland within each county. The livestock counts and pasture areas were used to determine livestock densities (e.g., number of cows per acre of pastureland) for each county, assuming livestock are evenly distributed over pasture area in the county.

The area of pastureland in each subwatershed and within each county was determined using GIS data layers. The pasture area of the subwatershed within each county and the livestock density for the counties were used to calculate the livestock counts within the portion of the subwatershed intersecting that county. That is to say, each county has a unique livestock density that was applied to the portion of the subwatershed within that county. The county/subwatershed livestock estimates were then summed to determine livestock counts for the entire subwatershed. The following example presents the calculation of beef cattle grazing in the West Fork 1 subwatershed of the Laurel River watershed. The county densities for beef cattle are

$$\text{Gunston County density} = \frac{2{,}850 \text{ beef cows}}{16{,}485 \text{ acres pastureland}} = 0.17\ \frac{\text{beef cows}}{\text{acre of pastureland}}$$

$$\text{Putnam County density} = \frac{6{,}376 \text{ beef cows}}{19{,}811 \text{ acres pastureland}} = 0.32\ \frac{\text{beef cows}}{\text{acre of pastureland}}$$

The West Laurel 1 subwatershed of the Laurel River watershed has 37 acres of pastureland in Gunston County and 172 acres of pastureland in Putnam County. Therefore, the total number of beef cows in the West Laurel 1 subwatershed is

$$\left(37 \text{ acres} \times 0.17\ \frac{\text{cows}}{\text{acre}}\right) + \left(172 \text{ acres} \times 0.32\ \frac{\text{cow}}{\text{acre}}\right) = 61 \text{ beef cows}$$

Based on local knowledge, it was assumed that cows spend 25 percent of their time confined and 75 percent of their time in pastures. For calculation purposes, “percent of time” is equivalent to “percent of cows.” Therefore the number of cows in the pasture and in confinement are

61 beef cows x 0.25 = 15.3 beef cows in confinement

61 beef cows x 0.75 = 45.8 beef cows grazing in pastures

Manure produced by cows in confinement is collected and spread on cropland within the watershed. ASAE (1998) provided manure production estimates and fecal coliform content of manure for various agricultural animals, including beef cows. According to ASAE, Beef cows produce an estimated 14,400 grams of manure a day with a fecal coliform content of 4.85×10^6 counts per gram. Therefore the amount of manure (and fecal coliform) available for application to watershed cropland is

$$15.3 \text{ cows} \times 14{,}400\ \frac{g}{\text{day} \cdot \text{cow}} = 220{,}320\ \frac{\text{grams of manure}}{\text{day}}$$

$$220{,}320\ \frac{g}{\text{day}} \times 4.85x10^6\ \frac{\text{fecal coliforms}}{g} = 1.07x10^{12}\ \frac{\text{fecal coliforms}}{\text{day}}$$

Based on local knowledge, it is assumed that 50 percent of the cows in the pasture have access to streams for watering and that cows with access to streams spend 25 percent of their time in the water.

45.8 cows x 0.50 x 0.25 = 5.7 cows in the water at any time

Assuming the fecal coliform production rate for beef cows provided in ASAE (1998), the load contributed directly to watershed streams by watering beef cattle is

$$5.7 \text{ cows} \times 14{,}400\ \frac{g}{\text{day} \cdot \text{cow}} \times 4.85x10^6\ \frac{\text{fecal coliforms}}{g} = 3.98x10^{11}\ \frac{\text{fecal coliforms}}{\text{day}}$$

The number of grazing cattle that are not watering and potentially contributing bacteria loads to the pasture surface is

45.8 cows in pasture - 5.7 cows watering = 40.1 cows grazing

The fecal coliform load contributed to the land surface by grazing cattle is

$$40.1 \text{ cows} \times 14{,}400\ \frac{g}{\text{day} \cdot \text{cow}} \times 4.85x10^6\ \frac{\text{fecal coliforms}}{g} = 2.8x10^{12}\ \frac{\text{fecal coliforms}}{\text{day}}$$

Example: Fecal Indicator Loading Assessment for a Feedlot

The example site is a 5-acre unpaved beef cattle feedlot in a plains state, with a normal summer-fall population of 800 cattle. The site is on silt loam soil and about 1/4 mile from a perennial stream with a baseflow (i.e., return flow from ground water) of 5 ft^3/s. Based on local climate, soil type, and soil condition, local agricultural agencies estimate that 50 percent of summer-fall precipitation will exit the site as surface runoff. Over the July to November period, the average runoff is 2.5 inches per month, occurring in an average of 4 runoff events per month, with an average duration of 4 hours. Average runoff per runoff event is thus 2.5 in./4 = 0.625 in. per event.

Local experience also indicates that a good order-of-magnitude estimate of fecal coliform concentration in runoff from a feedlot of this type with this number of animals is 5×10^7 CFU/100 mL. Not all fecal indicator bacteria washed off the site will reach the stream. The site has few BMPs in place, however, and the local agricultural agencies estimate that about 80 percent of the coliform load leaving the site will reach the stream. The concentration delivered to the stream is thus reduced to 80% x $5 \times 10^7 = 4 \times 10^7$ cfu/100 mL.

The first step in the source loading analysis is estimating an approximate loading rate during runoff events. This is accomplished with a simple mass balance using average flow and concentration. First, calculate the total flow volume from the feedlot during an average event:

$$0.625\ \frac{in.}{event} \cdot \frac{1\ ft}{12 in.} \cdot 5\ acres \cdot 43{,}560\ \frac{ft^2}{ac} = 11{,}344\ \frac{ft^3}{event}$$

Next, calculate the flow rate from the feedlot during an event:

$$11{,}344\ \frac{ft^3}{event} \cdot \frac{1\ event}{4\ hours} \cdot \frac{1\ hour}{60\ min} \cdot \frac{1\ min}{60\ sec} = 0.8\ \frac{ft^3}{sec}$$

The concentration in the receiving stream is easily calculated by the following equation:

$$C_{stream} = \frac{(Q_{feedlot})(C_{feedlot}) + (Q_{background})(C_{background})}{(Q_{feedlot}) + (Q_{background})}$$

where Q is the flow in ft^3/s and C is the concentration in cfu/100 mL. Assuming a flow at base levels of 5 ft^3/s and a background concentration in the stream of 15 cfu/100 mL, the resulting instream concentration is:

$$\frac{(0.8\ ft^3/s)(4\ x\ 10^7\ cfu/100\ mL) + (5\ ft^3/s)(15\ cfu/100\ mL)}{(0.8\ ft^3/s)(5\ ft^3/s)} = 5.5\ x\ 10^6\ \frac{cfu}{100\ mL}$$

Although this concentration is high, it is expected to occur infrequently during wet weather, with an average of 4 runoff events per month. Although the rain event lasts 4 hours, the resulting concentration is assumed to represent the daily concentration. The background concentration in the stream is 15 cfu/100 mL and is expected to occur on the other 26 days per month. Given these assumptions, the monthly geometric mean in the receiving stream is estimated as:

$$\sqrt[30]{(15)^{26}\ x\ (5.5x10^6)^4} = \sqrt[30]{3.47x10^{57}} = 82.80\ \frac{cfu}{100mL}$$

Recommendations: Source concentrations of fecal indicators are often region-specific; therefore, site or region-specific monitoring data should always be used when available. Begin with a simple approach to fecal indicator loading estimation by always starting with the assumption that a model is not required. Depending on the complexity of the combined fecal indicator sources in the watershed, loads can be estimated easily by conducting a literature search or more complexly when necessary, by using a combination of long-term monitoring and modeling. For point sources, estimation of loads should be obtained from effluent monitoring reqired for NPDES permits. Nonpoint sources are more difficult to estimate; therefore, models for flow volume and pollutant concentration in runoff can be used. The concentration in runoff can be obtained through a calibration exercise against measured in-stream data, but once again, the best estimates come from site-specific monitoring. Urban storm water loads can be estimated using a variety of techniques from simple loading rate assumptions and constant concentration estimates from literature values (as with the FecaLOAD model), to more complex models that are capable of accounting for pathogen die-off. Local health departments often have information on failing septic systems, but if not, a literature value for failure rate can be multiplied by the number of septic systems in the area. Rural nonpoint loads can also be estimated through various levels of complexity. These various approaches range from a simple assignment of estimated average rate of pathogen loading to a given land use from site-specific analysis to a more detailed model such as AGNPS, which accounts for temporal and spatial variability. Try to obtain information on rural nonpoint loading rates from sources such as local agencies or watershed and university studies. When literature values or site-specific values are not available for a particular source, similar source values can be substituted (i.e., raw sewage for septic effluent). The use of common raw sewage concentrations as an alternate value results in the incorporation of an implicit margin of safety into the analysis because of the potential for overestimation of the actual indicator bacteria concentration.

RECOMMENDATIONS FOR SOURCE ASSESSMENT

Using all available information, develop a comprehensive list of the potential and actual pathogen sources to the waterbody of concern. Develop a plan for identifying and accounting for the load originating from the identified sources in the watershed.

- Use GIS or maps to document the location of sources and the processes important for delivery to the waterbody.
- Identify all government agencies and nongovernment organizations active in the watershed, and conduct interviews and collect information.
- Group sources into some appropriate and manageable unit (e.g., by delivery mechanism, location) for evaluation using the available resources and analytical tools.
- Ideally, monitoring data should be used to estimate the magnitude of loads from various sources. In the absence of such data, some combination of literature values, best professional judgment, and appropriate empirical techniques/models is necessary. In general, the simplest approach that provides meaningful predictions should be used.

RECOMMENDED READING

(Note that a ful list of references is included at the end of this document.)

Donigian, A.S. Jr. and W.C. Huber. 1991. *Modeling of Nonpoint Source Water Quality in Urban and Non-urban Areas*. EPA/600/3-91/039. U.S. Environmental Protection Agency, Athens, GA.

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Novotny, V., and H. Olem. 1994. *Water Quality: Prevention, Identification, and Management of Diffuse Pollution*. Van Nostrand Reinhold, New York.

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USEPA. 1992a. *A Quick Reference Guide: Developing Nonpoint Source Load Allocations for TMDLs*. EPA 841-B-92-001. U.S. Environmental Protection Agency, Washington, DC.

USEPA. 1997b. *Compendium of Tools for Watershed Assessment and TMDL Development*. EPA841-B-97-006. U.S. Environmental Protection Agency, Office of Water, Washington, DC.

Linkage Between Water Quality Targets and Pollutant Sources

Objective: Define a linkage between the selected water quality targets and the identified source(s) to identify total assimilative capacity for pathogen or indicator bacteria loading or total load reduction needed.

Procedure: Determine the cause-and-effect relationship between the water quality target and the identified source(s) through data analysis, best professional judgment, models, and/or previously documented relationships. Use this linkage to determine what pathogen loads or conditions are acceptable to achieve the desired level of water quality. Develop approaches for determining an appropriate margin of safety.

OVERVIEW

One of the essential components of developing a TMDL is to establish a relationship (linkage) between source loadings and the numeric indicators chosen to measure the attainment of uses. Once this link has been established, it is possible to determine the capacity of the waterbody to assimilate fecal indicator loadings while still supporting its designated uses. Based on this analysis, allowable loads or needed load reductions can be allocated among the various pollutant sources. The link can be established through a range of techniques, from the use of qualitative assumptions backed up by sound scientific justification to the use of sophisticated modeling techniques. Ideally, the link can be based on a long-term set of monitoring data that allows the TMDL developer to associate certain waterbody responses to flow and loading conditions. More often, however, the link must be established by using a combination of monitoring data, statistical and analytical tools (including simulation models), and best professional judgment.

Key Questions to Consider for Linkage Between Water Quality Targets and Pollutant Sources

1. What type of analysis is appropriate for linking the water quality target(s) and identified pollutant sources?
2. What are the basic components of analysis for linking water quality targets and sources?
3. What complicating factors can influence the linkage analysis?

This section recommends appropriate techniques for establishing a source-indicator link. As with the prediction of pollutant source loadings, the analysis can be conducted using methods ranging from simple to complex.

KEY QUESTIONS TO CONSIDER FOR LINKAGE BETWEEN WATER QUALITY TARGETS AND POLLUTANT SOURCES

1. What type of analysis is appropriate for linking the water quality target(s) and identified pollutant sources?

Before choosing an appropriate method for linking instream water quality targets and sources, qualitative assumptions can be used to develop a screening-level linkage between sources and water quality targets. Qualitative assumptions must be backed up by sound scientific justification and thorough literature reviews. These assumptions can be a starting point in the linkage process.

Analytical methods appropriate for linking water quality targets and sources can use empirical approaches based on observed information, simple approaches, screening-level model analysis, or detailed modeling. TMDLs can incorporate one or more of these approaches to characterize the linkage between a target and source loading.

Empirical approaches

Empirical approaches use existing data to determine the linkage between sources and water quality targets. Bacteria indicators for pathogen TMDLs are relatively easy and inexpensive to monitor, and many states have extensive databases from coliform monitoring below known sources such as WWTPs and CSOs. In some cases, it might be appropriate to address the linkage between loading and exposure concentrations empirically, by comparing historical records of load and corresponding exposure concentrations. If sufficient observations are available to characterize the relationship between loading and exposure concentration across a range of loads, this information could be used to establish the linkage directly, using, for example, a regression approach.

Simple Approaches

Permitted point sources are required to meet water quality standards for indicator bacteria at the point of discharge or edge of the mixing zone, as specified in the state water quality standard. Simple dilution calculations and/or compliance monitoring (for existing discharges) are often adequate for this task. Compliance with end-of-pipe or mixing zone requirements establishes a baseline for developing a TMDL. Wasteload allocations for point sources should first be evaluated to determine if existing limits are adequate. If these controls are inadequate to meet standards (due, for instance, to the combined loading of multiple point sources), additional reductions in allocations will be needed to achieve the loading capacity.

Screening-level model analysis and detailed modeling

In cases where pathogen sources are complex and subject to influences from physical processes, an analysis of fate and transport might be needed to establish the linkage, typically using water quality models. A modeling approach can incorporate analysis of fate and transport issues such as mixing zone considerations, die-off rates, consideration of advection and dispersion, and influence of external factors, such as insolation, on die-off rates. Modeling techniques can vary in complexity, using one of two basic approaches—steady-state or dynamic modeling. Steady-state models use constant inputs for effluent flow, effluent concentration, receiving water flow, and meteorological conditions. Generally, steady-state models provide very conservative results when applied to wet weather sources. Dynamic models consider time-dependent variation of inputs. A daily averaging time is suggested for bacteria. The two modeling approaches are listed in order of increasing complexity as follows:

1. Steady-state analysis, in which a design condition of maximum impact (maximum load, low dilution capacity, low die-off rate) is selected and some interpretation of frequency is added.
2. Dynamic modeling, in which the analysis attempts to simulate the actual frequency of exposure concentrations.

Typically, a scoping analysis using empirical analysis and/or steady-state modeling is used to review and analyze existing data as a first step prior to any complex modeling. Scoping helps formalize the objectives of the process and provides a guide to what type(s) of detailed modeling, if any, might be appropriate.

2. What are the basic components of analysis for linking water quality targets and sources?

Identify targets

As described in Section 4 of this protocol, Identification of Water Quality Indicators and Target Values, the indicator for pathogen TMDLs may be a numeric water quality criterion (e.g., fecal coliform count) or some surrogate measure developed to protect the designated or existing use (e.g., recreation).

Quantify sources

To what degree are sources known and quantified? Are all significant sources of a given pollutant contributing to water quality impairment known? If not, what other potential sources should be considered? Determining the relative contributions of different sources to waterbody impairment is also important to subsequent analyses. Quantifying sources is addressed in Section 5, Source Assessment.

Locate critical points

If a watershed has many impaired segments, how is an analysis constructed to demonstrate that WQSs will be attained throughout the watershed? That is, where should the analysis focus? Monitoring or simulating fecal indicator concentrations at every point throughout the watershed is often not practical. Instead, the scoping effort should be targeted to areas where the waterbody is most sensitive to impacts from loads (critical points). Where only point sources to a river and a single water quality standard are concerned, the edges of the mixing zone below discharges are obvious critical points. More often, though, there are significant nonpoint sources or estuarine mixing, making the determination more difficult. Finally, more restrictive standards might apply to areas downstream of discharges, such as recreation areas, which become additional critical points for analysis.

Identify critical conditions

Under what conditions is a waterbody most likely to exceed water quality standards? When will it be most difficult for it to achieve water quality targets? Understanding when a waterbody is most vulnerable to adverse impacts is necessary for deciding on a design condition or conditions for the linkage analysis. This means thoroughly understanding the effects of dilution, temperature, load timing, and sometimes other factors on pollutant impacts. Ideally, the design condition will identify the combination of environmental factors that result in meeting the water quality criterion and will have an acceptably low frequency of occurrence.

Continuous loading sources (e.g., point sources) often most greatly impact water quality under low-flow, dry-weather conditions, when dilution is minimal (USEPA, 1991a). Typically, the lowest in-stream flows occur in summer or early fall when in-stream temperatures are high. However, high temperatures promote more rapid pathogen die-off.

In contrast, intermittent or episodic loading sources (e.g., surface runoff) that are rain-related can have serious water quality impacts under various flow conditions. Sometimes, maximum impacts from episodic loading occur at high flows instead of at low flows. For example, the elevated spring flows associated with snowmelt can contain high concentrations of fecal bacteria, especially when snowmelt originates from agricultural areas where manure is spread in winter or from urban areas where residents practice poor pet curbing. Consider also a more complex case in which a small tributary delivers fecal indicator bacteria to a river. The river's pathogen load is positively, although not linearly, correlated with flow in the higher-order stream. (Both waters respond to regional precipitation patterns.) The in-stream concentration from the tributary load will be affected by the competing influences of increased load and increased dilution capacity, resulting in a peak impact at some flow greater than baseflow. If a point source was also present, a dual design condition might be necessary.

Appropriate critical design conditions for an analysis should not exceed the frequency of occurrence limit stated in the water quality standard. For instance, to approximate the geometric mean coliform count, as measured over a 30-day period, an appropriate critical design condition for flow might be the minimum geometric mean 30-day flow. (As noted above, design conditions might need to be determined simultaneously for flow, temperature, and other factors.)

Simple, scoping-level modeling, coupled with empirical (graphical and statistical) data analysis, can usually address questions raised in this step. Scoping modeling typically involves simple, steady-state analytical solutions (e.g., exponential decay models for bacterial die-off) for a rough, first-cut analysis of the problem. These scoping analyses are not expected to provide highly accurate, quantitative answers, particularly when episodic wet-weather loads are involved. However, they can provide a valuable preliminary approximation of relative impacts, which is essential for focusing the subsequent analysis.

USEPA (1988) discusses methods for evaluating multiparameter design conditions from observations. Procedures for the implementation of state water quality standards may provide information to guide the determination of design conditions, as well.

Evaluate need for more complex analyses

Are the simplifying assumptions of the scoping analysis likely to bias results? For instance, if the effect of an episodic load is approximated by using a steady-state model, how is the actual impact likely to differ from the scoping prediction, which does not take into account the interaction of pollutant load and runoff flow, presence of concentration spikes, and other factors? Identifying sources of bias is crucial to determining the need for more complex modeling approaches.

3. What complicating factors can influence the linkage analysis?

Fecal indicator considerations, statistical variability of fecal indicator standards, mixing zone considerations, and pathogen die-off rates are important factors that help to shape the linkage analysis.

The linkage analysis may address a number of complex factors, including the following:

- Nonpoint sources, such as runoff from animal operations, might be a significant source of bacteria in the watershed.
- Transport via groundwater from septic tanks and waste lagoons might be important in some settings.
- The typical water quality target of fecal coliform count might not be a good indicator for certain disease vectors with significantly longer survival in the environment than coliforms, such as *Cryptosporidium* oocysts.
- Die-off rates can be affected by a wide variety of environmental factors, including temperature, light, salinity, and others, as discussed below.

Indicator considerations

The way in which the linkage is evaluated will depend on how the fecal indicator target value for the analysis and type of use is specified. The fecal indicator target value might have been selected based on a direct evaluation of risk to human health. More typically, however, fecal indicator TMDLs are based on meeting state ambient water quality standards for indicator bacteria. In many cases, fecal indicator TMDLs will also need to address areas of special concern where the probability of pollutant exposure is higher and stricter standards might apply. These include features such as drinking water intakes, public beaches and other areas of contact recreation, and shellfishing beds.

Statistical variability of indicator standards

Indicator bacteria standards are written in many different ways. A typical freshwater standard for contact recreation is a dual form standard that specifies that the 30-day geometric mean of *E. coli* counts is not to exceed 126 per 100 mL (on a minimum of five samples) and the single sample maximum allowable density for a designated beach area is 235 *E. coli* counts per 100 mL. Further, a 30-day average might be difficult to predict in the presence of variable hydrology or significant loading from episodic events such as CSOs. Therefore, it is also important to give some attention to the frequency or statistical aspects of evaluating the linkage for bacterial indicator TMDLs.

Mixing zone considerations

Compliance with water quality standards at the edge of a specified mixing zone depends on how well an effluent mixes with its receiving water. The degree of mixing depends on how the discharge is configured, as well as the character of the receiving water and effluent. For example, coliform concentrations resulting from a discharge that mixes rapidly with the entire cross-sectional area of a river are expected to be much lower than if the same discharge mixed slowly with only a portion of that cross-sectional area. Also, an effluent that is warmer or less saline than the receiving water will tend to be buoyant and rise or float on, rather than mix with, the receiving water in the vicinity of the outfall. It is also important to consider water quality standards implementation procedures which limit the size of the mixing zone (e.g., 20 percent of the 7Q10 flow).

These near-field analyses are addressed by mixing zone models. Mixing analyses are particularly important for estuaries and stratified lakes, where the advective energy available for mixing may be less than that in rivers and buoyancy differences are likely to be important. To address mixing in estuaries, USEPA developed the CORMIX expert systems methodology. This model and other techniques for modeling the mixing process are discussed in Jirka (1992). Representation of mixing in waterbodies of all types is also discussed in detail in Fischer et al. (1979).

Outside the initial mixing zone, transport of bacteria is usually described as a laterally mixed process in rivers and narrow reservoirs. More complex two- or three-dimensional models may be needed for estuaries and lakes, where vertical mixing is more significant to pathogen and fecal indicator die-off and transport than lateral mixing. Whether modeling in one, two, or three dimensions, a key to predicting far-field bacteria concentrations is accurately representing natural die-off or decay of bacteria in the environment.

Pathogen die-off rates

Fecal indicators and pathogenic organisms typically have a limited ability to survive outside their hosts. A large number of factors govern the survival of pathogenic organisms in waterbodies. Indicator bacteria die-off is considered to be best represented by a first-order equation (Tsonis, 1992; Thomann and Mueller, 1987). The overall first-order decay rate, K_B (day^{-1}) can be written in the following form (Thomann and Mueller, 1987):

$$K_B = K_{B1} + K_{B1} + K_{Bs} - K_a$$

where K_{B1} = basic death rate as a function of temperature, salinity, and predation;
K_{B1} = death rate due to sunlight;
K_{Bs} = net loss (gain) due to settling (resuspension); and
K_a = aftergrowth rate.

In practice, however, it has often been judged sufficient to approximate the die-off of organisms with a simple first-order or exponential assumption, which states that the rate of loss is proportional to the concentration. This alternate way of expressing the overall decay rate describes the decline of bacteria in the time it takes to obtain 90 percent mortality of the original number of bacteria assuming a first-order loss. The 90 percent mortality time, T_{90}, is given by the equation

$$0.10 = \exp(-K_B T_{90})$$

or

$$T_{90} = 2.3/K_B$$

Die-off equations may be applied sequentially to a series of stream reaches with point source inputs. Analytical solutions are also available for streams with distributed nonpoint sources arising from tributary inflow (Mills et al., 1985) and for distributed input mobilized by sedimentation and scour (Thomann and Mueller, 1987). For estuaries and lakes where mixing or dispersion is important, a dispersion coefficient is included in the solution, as well as calculations distinguishing between up- and down-estuary flow. The equations above can be modified to take into account specific factors influencing die-off, such as temperature or insolation. For more detailed and mathematical discussions of die-off equations and calculations, see Bowie et al. (1985), Mills et al. (1985), and Thomann and Mueller (1987).

In general, for the exponential decay approach, fecal indicator bacteria can be modeled like any other constituent assumed to exhibit exponential decay. Many different water quality models (both analytical formulations and computerized models) can be used to represent fecal indicator bacteria as well as other bacteria indicators. For instance, QUAL2E allows direct input of a coliform bacteria concentration, with temperature-dependent exponential decay. In WASP/TOXI5, fecal indicator bacteria are not discussed directly; however, they can be simulated by specifying coliform bacteria as a "chemical" with an appropriate exponential biodegradation rate. Other potential models include CE-QUAL-RIV1, CE-QUAL-W2, and HSPF/BASINS. A discussion on receiving water models is included in the *Compendium of Tools for Watershed Assessment and TMDL Development* (USEPA, 1997b).

Factors influencing pathogen die-off rate

Many environmental parameters influence the die-off, fate, and distribution of fecal indicator bacteria in waters. The major factors that influence the kinetic behavior of disease pathogens after discharge to a waterbody are (Thomann and Mueller, 1987):

- Sunlight
- Temperature
- Salinity
- Predation
- Nutrient deficiencies
- Toxic substances
- Settling of the organism population after discharge
- Resuspension of particulates with associated sorbed organisms
- Aftergrowth, that is, the growth of organisms in the body of water

A more detailed discussion of the factors influencing mortality is contained in Bowie et al. (1985). Of these factors, temperature is the most widely considered. It is usually represented by an approximate form that relates mortality at 20 °C (K_{20}) to mortality at any other temperature (K_T). Even when normalized to K_{20}, values of coliform disappearance rates vary widely. Bowie et al. (1985) summarize disappearance rates used in a variety of modeling studies, ranging from 0.01 to 8.0 per day at 20 °C. Various models have been advanced to account for some of the other factors that cause the exponential decay rate to vary. For instance, Mancini (1978) provides a model for the incorporation of

Example: Calculation of Exponential Decay of Coliform Concentration

A WWTP discharges to a river with an average coliform concentration of 200 organisms per 100 mL. The decay coefficient for ambient temperature conditions is 0.5 per day. Stream velocity is 0.2 ft/s, equivalent to 3.3 mi/d. At a point 1 mile downstream, the expected concentration is given by

$$C = C_0 \exp(-KX/U)$$

where C = concentration of fecal indicator bacteria,
K = decay coefficient,
X = distance along axis of flow, and
U = flow velocity.

The expected concentration (organisms/100 mL) is:

$$C = 200 \cdot \exp\left(-\frac{0.5}{day} \cdot \frac{1\ mi}{3.3\ mi/day}\right) = 172\ \frac{organisms}{100\ mL}$$

salinity, temperature, and solar radiation into estimation of the mortality rate. In practice, because the first-order die-off assumption is itself a gross approximation, there are limits to the accuracy that can be attained by these prediction methods. It is clear from many studies that populations of coliforms and other fecal indicators typically include susceptible subpopulations, which die off quickly in the environment, and more resistant strains, which die off more gradually. The result is that die-off tends to slow down below the rate predicted by an exponential fit to the first few days. Because of the complexity of parameters affecting die-off, it is always advisable to examine site-specific die-off rates.

Die-off rates are particularly problematic for the infectious cysts and oocysts of *Giardia lamblia* and *Cryptosporidium parvum*. Apparently, these can survive significantly longer in the environment than fecal coliform bacteria. It does appear, however, that die-off rates for these organisms exhibit a significant temperature dependence, so methods similar to those used for coliform bacteria can be used to adjust for temperature variability. Information on die-off rates will likely improve as current monitoring programs progress and additional data are collected.

Fecal indicator standards are typically written as geometric means. A full evaluation of the linkage thus needs to address the *statistical distribution* of the resulting concentrations. Prediction that a water quality standard will be achieved (or not achieved) under a given set of conditions is not necessarily informative as to whether the geometric mean count will be achieved.

Even when a point source has constant flow and constant fecal indicator load, resulting exposure concentrations in the environment will vary with time because of continually varying flow and dilution capacity in the receiving waterbody. Other factors that influence coliform survival, such as temperature, also will vary in time. Analysis is made more difficult when significant nonpoint sources are present. Precipitation-driven loads are likely to be at their highest when dilution flows in the receiving water are also elevated since both respond to precipitation. Examples of some fecal indicator and pathogen die-off rates are shown in Table 6-1.

Types of dynamic receiving water models

USEPA (1991b) recommends three dynamic receiving water modeling techniques to be used when an accurate estimate of the frequency distribution of projected receiving water quality is required—continuous simulation, Monte Carlo simulation, and lognormal probability modeling.

Continuous simulation models combine daily (or other time step) measurements or synthesized estimates of effluent flows, effluent loads, wet-weather source concentrations/loads, and receiving water flows to calculate receiving water concentrations. A

Table 6-1. Examples of fecal indicator and pathogen die-off rates

Organism	K_B (day^{-1})[a]	Remarks	Reference
Coliform			
Total coliform	1-5.5	Freshwater: 20°C	Thomann and Mueller, 1987
	0.7-3.0	Seawater: 20°C	Thomann and Mueller, 1987
	0.42	Nonsterile river water (12 days) (no temperature indicated)	Baudisova, 1997
Fecal coliform	37-110	Seawater, sunlighted	Thomann and Mueller, 1987
	0.51	Non sterile river water (12 days) (no temperature indicated)	Baudisova, 1997
	0.043 0.124 0.146	Sand: 4 °C 25 °C 35 °C	Howell et al., 1996
	0.043 0.108 0.156	Loam: 4 °C 25 °C 35 °C	Howell et al., 1996
	0.025 0.022 0.083	Clay: 4 °C 25 °C 35 °C	Howell et al., 1996
	0.010-0.023	Sediment at 8 °C	Sherer et al., 1992
E. coli	0.08-2.0	Seawater, 10-30 0/00	Thomann and Mueller, 1987
	0.53	Nonsterile river water: 37°C (12 days)	Baudisova, 1997
	0.54	Nonsterile river water: 44°C (12 days)	Baudisova, 1997
	0.102	Natural surface water: 5°C (42 days)	Medema et al.,1997
	0.202 0.049	Natural surface water: 15°C[b] (0-14 days) (14-42 days)	Medema et al.,1997
Fecal streptococci			
Streptococcus faecalis	0.4-0.9	Freshwater: 20°C	Thomann and Mueller, 1987
	0.1-0.4	Freshwater: 4°C	Thomann and Mueller, 1987
	0.3	Storm water: 20°C (0-3 days)	Thomann and Mueller, 1987
	0.1	Storm water: 20°C (3-14 days)	Thomann and Mueller, 1987
Streptococcus bovis	1.5	Storm water: 20°C	Thomann and Mueller, 1987

Table 6-1. Examples of fecal indicator and pathogen die-off rates (continued)

Fecal streptococci	18-55	Seawater, sunlighted	Thomann and Mueller, 1987
	0.013 0.119 0.103	Sand: 4 °C 25 °C 35 °C	Howell et al., 1996
	0.010 0.064 0.130	Loam: 4 °C 25 °C 35 °C	Howell et al., 1996
	0.019 0.025 0.095	Clay: 4 °C 25 °C 35 °C	Howell et al., 1996
	0.018-0.033	Sediment at 8 °C	Sherer et al., 1992
Fecal enterococci	0.077	Natural surface water: 5 °C (42 days)	Medema et al.,1997
	0.233 0.025	Natural surface water: 15 °C[b] (0-14 days) (14-42 days)	Medema et al.,1997
Pathogens			
Salmonella typhimurium	1.1	Storm water: 20 °C (0-3 days)	Thomann and Mueller, 1987
	0.1	Storm water: 20 °C (3-14 days)	Thomann and Mueller, 1987
Cryptosporidium	0.010	Natural surface water: 5 °C (35 days)	Medema et al., 1997
Cryptosporidium	0.024	Natural surface water: 15 °C (35 days)	Medema et al., 1997
Viruses			
Coxsackie	0.12	Marine waters: 25 °C	Thomann and Mueller, 1987
	0.03	Marine waters: 4 °C	Thomann and Mueller, 1987
Echo 6	0.08	Marine waters: 25 °C	Thomann and Mueller, 1987
	0.03	Marine waters: 4 °C	Thomann and Mueller, 1987
Polio type 1	0.16	Marine waters: 25 °C	Thomann and Mueller, 1987
	0.05	Marine waters: 4 °C	Thomann and Mueller, 1987

[a] K_B = the overall first-order decay rate.
[b] Biphasic die-off kinetics: phase 1: days 0-14, phase 2: days 14-42.

deterministic model is applied to a continuous time series of these variables so that the model predicts the resulting concentrations in chronological order with the same time sequence as the input variables. This approach enables a frequency analysis of concentrations at any given point of interest. The analysis automatically takes into account the serial correlation that may be present in flows and other parameters, as well as the cross-correlations between measured variables. Continuous simulation of fecal indicator bacteria can be undertaken in a variety of modeling packages, such as HSPF/BASINS.

Continuous simulation is potentially the most powerful method available for accurate prediction of the frequency of receiving water concentrations. However, it does have limitations. Continuous simulation, if applied to detailed analysis, can be data-intensive. Often, limited pollutant monitoring data are available (especially during storm events) to test the performance of the model.

Monte Carlo simulation models combine probabilistic and deterministic analyses. That is, this approach uses a deterministic water quality model with statistically described inputs. The model is run repeatedly, with each iteration randomly selecting input values from the input statistical distributions. If all the time-varying inputs (such as flows) and uncertain parameters are described statistically, the result is a simulated set of receiving water concentrations that reflects the statistical distribution of the model inputs; however, these concentrations will not follow the actual day-to-day sequence of real data. A particular strength of Monte Carlo methods is their ability to provide a direct assessment of model uncertainty by use of statistical representations of uncertain parameters and to provide an output distribution that allows specification of a percent likelihood that the water quality standard will be maintained. However, it is very difficult to incorporate realistic patterns of spatial and temporal cross-correlation between flows, loads, and other factors into the analysis.

USEPA has developed lognormal probabilistic dilution models to provide a simpler method of frequency analysis. For instance, in USEPA's DYNTOX model (USEPA, 1996b), the lognormal probabilistic approach takes a simple, deterministic stream dilution model, assumes that all the input parameters can be represented by lognormal distributions, and uses numeric integration to derive the resulting distribution of receiving water concentrations. Similar to a Monte Carlo analysis, the objective is to find the distribution of model predictions based on assumed distributions of loads, flows, and other factors. By making restrictive lognormal distribution assumptions, however, the problem can be solved directly, rather than by using the iterative procedure of the Monte Carlo method.

EXAMPLE: SCOPING THE LINKAGE FOR A CSO FECAL INDICATOR TMDL FOR AN ESTUARY

Of particular concern is fecal indicator contamination from CSOs and other sources that can introduce human waste and pathogens directly into a waterbody without treatment. In this example, a CSO discharging to an estuary causes intermittent impairment of designated or existing uses, including a shellfishery and contact recreation.

Identify indicator and water quality target

In this case, the water quality target is taken as the relevant state WQS. The coliform standard for this state is a dual form standard that specifies that the 30-day geometric mean of fecal coliform counts is not to exceed 200 per 100 mL (on a minimum of five samples) and not more than 20 percent of samples are to exceed 400 per 100 mL. A standard in this form does not specify a not-to-exceed count, and a 30-day average might be difficult to predict from episodic, irregularly spaced CSO events. How can a standard of this type be evaluated for episodic load? There are two basic approaches:
(1) attempt continuous simulation of a realistic series of CSO events, driven by historical rainfall records, predict daily concentrations, and compare the frequency of excursions to the WQS; and (2) take an approximate approach, which tries to ensure that the average loading over time meets the 30-day geometric mean standard and the maximum concentration meets the 20 percent criterion. The second approach is more appropriate for scoping the problem. Indeed, given the difficulties of obtaining an accurate simulation of CSOs, there is no guarantee that the more complex approach would yield more accurate results.

Quantify sources

Information on sources of contamination is a key to this example. CSOs are identified as the source of impairment. Their impact has not been rigorously proven, however, because exact loading is difficult to quantify and other sources of fecal coliforms may discharge to the estuary, including upstream storm water, agricultural runoff, and septic systems. Identified upstream sources are likely subject to substantial fecal indicator die-off before reaching the estuary. (With a travel time on the order of 2 days, a typical coliform mortality profile would reduce the original upstream load to around 10 percent.) Septic systems and potentially leaking sewer lines near the estuary might be significant sources. Total loading from sources upstream of tidal influence is best obtained from monitoring at the head of the estuary. Additional monitoring and data interpretation for the estuary are also needed to assess the relative importance of local septic systems and other sources in relation to the CSOs.

Locate critical points

Because coliform mortality is fairly rapid, concentrations are expected to decline away from the source. This implies that the standard should be enforced at the edge of the mixing zone, thus involving only a dilution or near-field analysis. Special-use areas such as beaches and shellfish beds might require additional attention and focus as another critical point, even if outside the mixing zone and subject to the same WQS. This example concentrates on impacts at a public beach 1.5 miles upstream from the CSO outfall.

Identify critical conditions

The critical conditions for scoping in this example reflected the dual nature of the WQS. Interpreting the 400 per 100 mL count as a not-to-be-exceeded target for the scoping (rather than an 80th percentile) provides a condition analogous to a design low-flow condition, which represents the minimum dilution capacity in the receiving water reasonably expected in conjunction with the episodic load. Dilution capacity and mixing processes are not expected to be strongly associated with the occurrence of CSO events in the estuary because (1) the tidal mixing component will always be present and (2) upstream flows are generated by a large watershed with a reasonable probability of being at low-flow conditions during a localized CSO loading event. Recommendations for design (critical dilution) conditions in estuaries are provided in USEPA (1991b, p. 74):

> In estuaries without stratification, the critical dilution condition includes a combination of low-water slack at spring tide for the estuary and design low flow for riverine inflow. In estuaries with stratification, a site-specific analysis of a period of minimum stratification and a period of maximum stratification, both at low-water slack, should be made to evaluate which one results in the lowest dilution.

Because this estuary did not exhibit strong stratification near the CSO outfall, unstratified critical dilution conditions apply. The low-water slack at spring tide and design low flow upstream are appropriate only at the point of the CSO discharge. At special points of concern farther away, the combination of reasonable flows and diffusion coefficients, which produces the maximum impact by combining relatively high rates of dispersive transport and relatively low dilution, must be evaluated. Finally, design conditions will also include temperature and salinity, both of which influence the coliform die-off rate.

For initial scoping, a steady-state analytical model for one-dimensional estuarine advection and dispersion was used. This solution is based on the assumption of an infinitely long estuary of constant area and is useful for estuaries that are sufficiently long to approach steady state near the outfall. The character of the solution is strongly controlled by the ratio KE/U^2, referred to as the estuarine number, which reflects the relative importance of diffusive and advective fluxes. As this number approaches zero, transport in the estuary becomes increasingly similar to river transport. In this estuary, the ratio is approximately 1.5, which indicates relatively strong tidal mixing with significant transport up-estuary.

For scoping, the geometric mean requirement of the WQS is taken as an average condition over time. That is, the 30-day time frame for this analysis is assumed to be long enough to allow the variability in the load, as well as tidal cycles, to be averaged out. The scoping, therefore, assumes a steady load in terms of an average over time. An advection-dispersion solution can again be used in this case. Another powerful scoping method for this type of case is the modified tidal prism method, which predicts pollutant concentration based on the observed average salinity profile in the estuary (Mills et al., 1985).

Results of the scoping analysis based on the one-dimensional advection-dispersion solution are shown in the box and Table 6-2. A mixing zone of one-half mile up-estuary and down-estuary of the outfall is allowed. The beach location, 1.5 miles up-estuary of the outfall, is of particular concern. The model was applied for a variety of conditions, including freshwater flows at 7Q10 and 30Q10 flows and loads at the estimated event maximum daily average load and long-term average load. Because the answer depends on the value assigned to the dispersion coefficient, sensitivity of the answer to dispersion coefficients was examined. Coefficients ranged from 2 to 3 square miles per day (mi^2/d), the expected range for the part of the estuary near the outfall.

It is most appropriate to compare the 200 per 100 mL standard to the 30Q10 upstream flow and average load (since the standard is written as a 30-day average) and the 400 per 100 mL standard to the 7Q10 upstream flow and event maximum load. Scoping indicates that the CSO can cause the short-term standard to be exceeded at the mixing zone boundaries and likely causes impairment of the up-estuary beach. Increasing the estimate of the dispersion coefficient increases the estimated concentration at the beach, reflecting increased up-estuary "smearing" of the contaminant plume, which illustrates that the minimum mixing power might not be a reasonable design condition for evaluating maximum impacts. WQS excursions at the beach are likely to occur only at low upstream flows, while the combination of average loads and 30Q10 fresh water flows is not predicted to cause impairment. In evaluating impacts at the beach, recall that scoping was conducted using a one-dimensional model that averages a cross-section. Even if the cross-sectional average is correctly estimated, impacts at a specific point (e.g., the beach) may be higher or lower than the estimated value, depending on tidal circulation patterns.

Scoping Assumptions for Estuarine CSO Example

Upstream Flows

7Q10 = 900 ft^3/s
U (7Q10) = 1.5 mi/d
30Q10 = 1500 ft^3/s
U (30Q10) = 2.5 mi/d

Estuary

A = 10,000 ft^2
E = 2–3 mi^2/d
T = 27 °C
K = 1.11/d
Unstratified

CSO

C = 1 x 10^6 coliform/100 mL
Q_e = 0.1 MGD average, 2 MGD maximum

where:

U = velocity
A = area
E = tidal macrodispersion coefficient
T = temperature
K = first-order decay coefficient
C = concentration
Q = flow rate

Evaluate need for more complex analyses

The scoping analysis suggests a strong probability of WQS excursions at the mixing zone boundary. The situation at the beach is less clear, since estimates depend strongly on the specified values of reasonable maximum loading and dispersion. The analysis at the mixing zone boundary alone might be sufficient to justify control of the source; it depends on the level of confidence in these estimates. For example, a first-cut

Table 6-2. Steady-state predictions of fecal coliform count in the estuary (organisms/100 mL)

Flow:	Upstream: 900 ft^3/s (7Q10)		Upstream: 1500 ft^3/s (30Q10)			
Load:	Event Maximum Load				Average Load	
Dispersion:	E = 2 mi^2/d	E = 3 mi^2/d	E = 2 mi^2/d	E = 3 mi^2/d	E = 2 mi^2/d	E = 3 mi^2/d
Mixing Zone, Upstream	838	821	596	651	30	33
Mixing Zone, Downstream	1212	1050	1102	981	55	49
Beach	252	333	123	207	6	10

estimate of the load required to maintain water quality standards could be specified for the period of time required to bring the combined sewer system (CSS) into accord with the nine minimum controls specified in USEPA's CSO control policy. When data on the effect of the minimum controls are collected, the next phase could involve more complex modeling and a more sophisticated wasteload allocation (Ambrose et al., 1992).

Keep in mind, however, that although detailed simulation of the estuary in two or more dimensions could provide more accurate results, it would be warranted only if interest in predicting transport to a specific point, such as the beach, is strong. An efficient strategy would be to implement initial CSS controls, estimate whether a residual problem is still likely at the beach, and proceed with more complex modeling only if the answer is unclear. (If it is clear there would still be a problem, complex modeling would not be needed to show that additional controls would be required.) Dynamic modeling approaches seek to develop a realistic estimate of the time series of WQS excursions resulting from episodic loads. Consequently, they attempt to estimate not just whether an excursion will occur, but at what frequency excursions of a given duration might be expected. This approach provides for a more sophisticated analysis of the actual risk posed by an episodic source. Estimation of the frequency of excursions of WQSs for waterbodies with wet-weather-dominated loading typically involves continuous simulation over a number of years of precipitation records. It is a logical way to proceed when sufficient resources are available to undertake such an analysis. However, continuous simulation is not always feasible because of a lack of data or constraints on available resources to perform the modeling analysis.

EXAMPLE: BACTERIAL LINKAGE ANALYSIS FOR TWO SOURCES TO A RIVER

An important aspect of linkage analysis is estimating the combined impact of two or more sources of fecal indicator loading. This is particularly challenging when episodic nonpoint sources are involved. This example addresses modeling the linkage between water quality impacts in a river and two sources of fecal indicator loading—a steady point source from a POTW and a dynamic, episodic nonpoint source in an urban separate storm sewer system.

The POTW has a design flow of 10 ft^3/s (6.5 MGD) and discharges to a small river with a median flow of 35 ft^3/s and a 7Q10 low flow of 8 ft^3/s. The climate is continental, with a dry summer and early fall. Maximum flows are associated with spring rain and snowmelt events. The POTW achieves varying rates of disinfection over the course of a year. During hot summer weather, the survival time for fecal coliform bacteria is shortened. The plant achieves an average concentration of 400 CFU/100 mL in effluent from July through September. In spring and fall the average concentration is 1,000 CFU/100 mL, and in winter the average concentration is 2,000 CFU/100 mL. Background fecal coliform concentration in the receiving stream is typically around 200 CFU/100 mL.

The discharge from an urban separate storm sewer system is located 3.5 miles *(x)* upstream. Average flow velocity in the river is 0.2 ft/s, or 3.28 mi/day *(u)*, so the average time of travel between the storm water discharge and the POTW is 1.07 days. Assuming a loss coefficient *(k)* of 1 day^{-1}, the average fraction of fecal indicator loading from the storm sewer still present at the POTW is:

$$e^{-kt} = e^{-kx/u} = e^{-\frac{1}{day} \cdot \frac{3.5mi}{3.28mi/day}} = 0.34$$

Significant discharge from the storm sewer system occurs about 20 to 30 times per year. Both flows and fecal indicator concentrations in the storm water are highly variable. The median fecal coliform concentration in storm water is 1,000 CFU/100 mL, but occasionally it might be an order of magnitude higher, particularly during first flush after dry periods and during snowmelt. Flow rates from the storm sewer system during individual events range up to 50 ft^3/s. Fecal indicator load input from the storm sewer system may occasionally result in a significant increase in in-stream concentrations at the point of the POTW discharge, depending on the dilution capacity available in the river.

The linkage analysis is conducted using a dynamic model to account for the dynamic nature of the episodic source. The dynamic model is, however, the simplest type conceivable since it is based on a simple one-dimensional steady-state mass balance mixing equation:

$$C_{mix} = \frac{C_{up} \cdot Q_{up} + C_w \cdot Q_w}{Q_{up} + Q_w} \tag{1}$$

where

C_{mix} = mixed concentration below a discharge;
C_{up} = concentration in the reach immediately above the discharge;
Q_{up} = flow above the discharge;
C_w = concentration in the discharge; and
Q_w = the flow in the discharge.

This equation is appropriate for estimating concentrations in a fully mixed cross section of a river with steady inputs. Any consistent units may be used in the equation. Although it is steady-state, it may be applied in a quasi-dynamic mode by using daily time series of upstream and discharge flows and concentrations (observed or predicted) to calculate a daily time series of mixed concentrations below the discharge. Mathematical representation of the linkage is completed through incorporation of a second equation, representing bacterial die-off via a first-order loss coefficient:

$$C_x = C_0 \cdot e^{-kx/u} \tag{2}$$

or

$$C_x = C_0 \cdot e^{-kt} \tag{3}$$

where

C_x = mixed concentration at a point a distance x below a discharge;
x = distance downstream from the discharge;
C_0 = mixed concentration immediately below a discharge;
k = first order loss coefficient;
u = flow velocity; and
t = travel time, x/u.

Equation (1) may first be used to examine the instream fecal indicator concentrations resulting from the POTW, plus natural background, at the point of mixing of the effluent. Using daily measured upstream flows and effluent flows and concentrations, expected instream concentrations over a typical year are shown in Figure 6-1. The jagged line represents the daily time series of mixed in-stream concentrations; the smoother, heavier line shows the moving geometric mean.

In-stream concentrations vary over the course of the year in response to a number of factors. During cold weather, the coliform removal efficiency is lower, resulting in greater loads. Concentration is highest in mid-winter, when in-stream flows are low. Concentration declines in February and March because of increased in-stream dilution capacity. Flows decline again in the summer, but removal efficiency also increases, resulting in low in-stream concentrations over the summer.

For this water, the state has specified a seasonal fecal coliform standard of 400 CFU/100 mL during the summer recreation season (May 1–October 15) and 1,000 CFU/100 mL during colder weather as a 30-day geometric mean. Both standards were generally met by the POTW effluent during this year, except for a brief period in October, although individual concentrations greater than 1,000 CFU/100 mL were observed.

What happens if the storm water effluent is also considered? Storm water can provide intermittent high loads; however, the impact is mitigated by the fact that storm water loading tends to occur when in-stream flows, and thus dilution capacity, are also higher. Figure 6-2 shows the sequence of daily concentrations with upstream stormflow included. As in Figure 6-1, both daily values and a 30-day moving geometric mean are shown.

The storm water flow has the potential to cause temporary high concentrations of fecal coliform bacteria in the river, primarily associated with the rising limb of storm water flow that carries the first flush of pollutants, and often occurs before a significant response in upstream flows. The geometric mean is, however, less responsive, unless a number of events occur in quick succession. This happens primarily during the spring runoff period. Here, the geometric mean concentrations

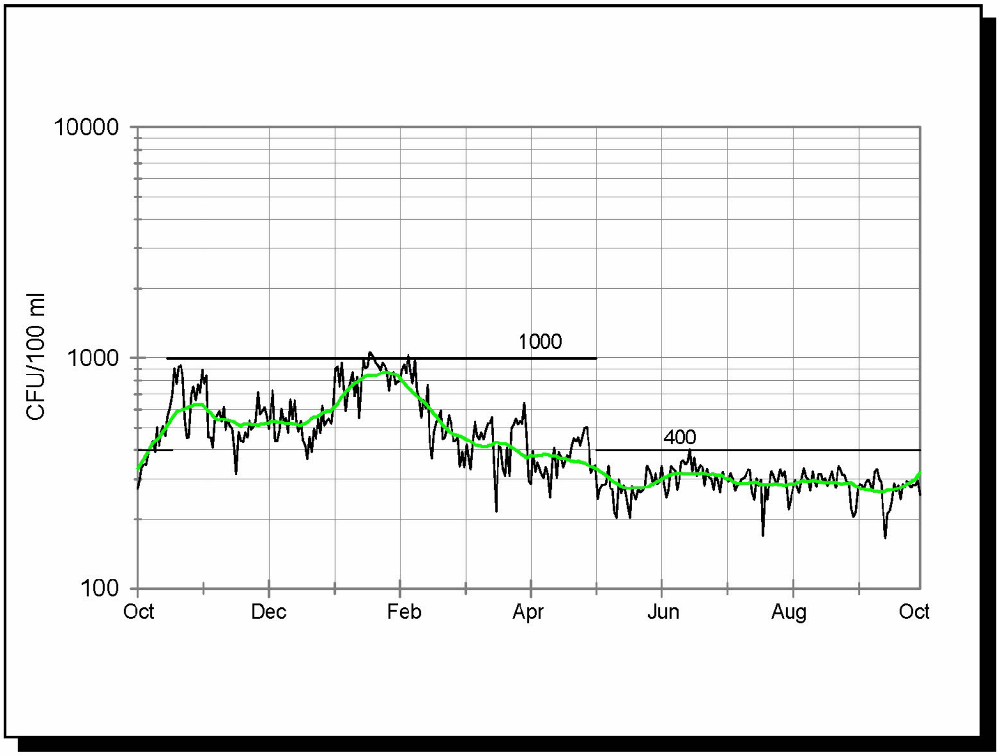

Figure 6-1. Daily in-stream fecal coliform concentrations resulting from POTW effluent. The jagged line shows the daily time series of mixed in-stream fecal coliform concentrations. The smooth line shows the moving geometric mean. 400 CFU/mL is the state-specified fecal coliform standard for the summer season (May 1–October 15). 1000 CFU/100mL is the cold weather standard.

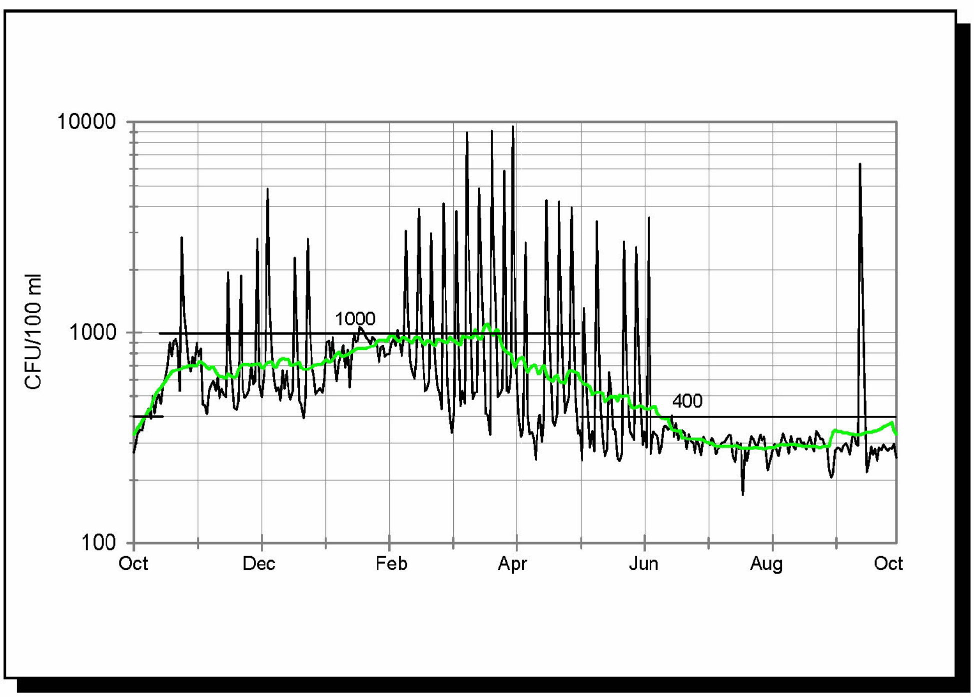

Figure 6-2. Daily in-stream fecal coliform concentrations resulting from POTW effluent plus storm water discharge. (See description under Figure 6-1.)

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are elevated substantially versus those calculated for the POTW effluent alone; however, because of the high dilution capacity in this period, the geometric mean concentrations generally remain below 1,000 CFU/100 mL.

This preliminary linkage analysis suggests that there are two periods in which excursions of the geometric mean standard for fecal coliform bacteria are likely to occur. These are early spring (March), in which frequent stormflow events may raise the in-stream geometric mean above the winter standard of 1000 CFU/100 mL, and late spring (May), when the warm weather standard of 400 CFU/100 mL comes into play but storm runoff events are still frequent. Only a single year is represented in Figures 6-1 and 6-2, and conditions may vary substantially from year to year. A more detailed linkage analysis might focus on the critical May time period, during which concentrations are likely to exceed standards and human exposure is likely. Simulation modeling could be used to examine expected concentrations across numerous years of May precipitation and flow.

RECOMMENDATIONS FOR LINKAGE BETWEEN WATER QUALITY TARGETS AND SOURCES

- Use all available and relevant data; ideally, the linkage will be supported by monitoring data, allowing the TMDL developer to associate waterbody responses with flow and loading conditions.

- Typically, a scoping analysis using empirical analysis and/or steady-state modeling can be used to review and analyze existing data prior to any complex modeling. A scoping analysis usually includes identifying targets, quantifying sources, locating critical points, identifying critical conditions, and evaluating the need for more complex analysis.

- When selecting a technique to establish a relationship between sources and water quality response, usually the simplest technique that adequately addresses all relevant factors should be used.

RECOMMENDED READING

(Note that a full list of references is included at the end of this document.)

Chapra, S. 1997. *Surface Water-Quality Modeling*. McGraw-Hill Publishers, Inc.

Thomann, R.V., and J.A. Mueller. 1987. *Principles of Surface Water Quality Modeling and Control.* Harper & Row, New York.

USEPA. 1997b. *Compendium of Tools for Watershed Assessment and TMDL Development*. EPA841-B-97-006. U.S. Environmental Protection Agency, Office of Water, Washington, DC. <http://www.epa.gov/owow/tmdl/techsupp.html>.

USEPA. 1988. *Technical Guidance on Supplementary Stream Design Conditions for Steady State Modeling. Technical Guidance Manual for Performing Waste Load Allocations*, Book VI, Chapter 2. U.S. Environmental Protections Agency, Office of Water, Washington, DC.

Allocations

Objective: Using total assimilative capacity developed in the linkage component, develop recommendations for the allocation of loads among the various point and nonpoint sources, while accounting for uncertainties in the analyses (i.e., margin of safety) and, in some cases, a reserve for future loadings.

Procedure: Determine the allocations based on determination of the acceptable loading (loading capacity), the margin of safety, and the estimated loads from all significant sources. The available load is then allocated among the various sources.

OVERVIEW

TMDLs are composed of the sum of individual wasteload allocations (WLAs) for point sources and load allocations (LAs) for both nonpoint sources and natural background levels for a given waterbody. The sum of these components must not result in the exceedance of water quality standards for that waterbody. In addition, the TMDL must include a margin of safety (MOS), either implicitly or explicitly, that accounts for the uncertainty in the relationship between pollutant loads and the quality of the receiving waterbody. Conceptually, this definition is denoted by the equation

$$\text{TMDL} = \Sigma\,\text{WLAs} + \Sigma\,\text{LAs} + \text{MOS}$$

For most pollutants, TMDLs are expressed on a mass loading basis (e.g., pounds per day). For fecal indicators, however, TMDLs can be expressed in terms of organism counts (or resulting concentration), in accordance with 40 CFR 130.2(i): "TMDLs can be expressed in terms of mass per time, toxicity, or other appropriate measure," and NPDES regulations at 40 CFR 122.45(f): "All pollutants limited in permits shall have limitations...expressed in terms of mass except... pollutants which cannot appropriately be expressed by mass."

Key Questions to Consider for Allocations

1. What are the steps involved for completing the allocations?
2. How should candidate allocations be evaluated?
3. How can TMDLs be translated into controls?
4. How should issues of equitability and fairness be addressed?
5. How should stakeholders be involved?

To establish a TMDL, the administering agency must find an acceptable combination of allocations that adequately protects water quality standards. However, deciding how to divide the assimilative capacity of a given watershed among sources can be a challenging task. Issues that affect the allocation process include:

- Economics
- Political considerations
- Feasibility
- Equitability
- Types of sources and management options
- Public involvement
- Implementation
- Limits of technology
- Variability in loads, effectiveness of BMPs

Although there is more than one approach to establishing TMDLs, typical steps in the process are addressed in following sections.

KEY QUESTIONS TO CONSIDER FOR ALLOCATIONS

1. What are the steps involved for completing the allocations?

The first step in establishing a TMDL is to specify the methods that will be used to incorporate an MOS. Section 303(d) of the CWA requires TMDLs to include "a margin of safety which takes into account any lack of knowledge concerning the relationship between effluent limitations and water quality." Given that TMDLs address both point source allocations (WLAs) and nonpoint source allocations (LAs), this concept may be extended to cover uncertainty in BMP effectiveness in addition to effluent limitations.

There are two basic methods for incorporating the MOS (USEPA, 1991a, 1999):

- Implicitly incorporate the MOS using conservative model assumptions to develop allocations, or
- Explicitly specify a portion of the total TMDL as the MOS; use the remainder for allocations.

In many cases, the MOS is incorporated implicitly. In such cases, the conservative assumptions that account for the MOS should be identified. An explicit calculation, including evaluation of uncertainty in the linkage analysis, has the advantage of clarifying the assumptions that go into the MOS determination.

2. How should candidate allocations be evaluated?

TMDLs by definition are combinations of WLAs and LAs that allocate assimilative capacity to achieve water quality goals, including a margin of safety and a consideration of seasonal variation. The first step in the evaluation is to determine which segments and sources require allocation adjustment to achieve water quality standards and designated or existing uses. The actual adjustment to allocations will likely be based on the administering agencies' policies and procedures. For instance, should reductions be spread out across all sources or apply to only a few targeted sources? Each agency may have its own criteria for making these decisions (e.g., magnitude of impact, degree of management controls now in place, feasibility, probability of success, cost, etc.) The following subsections provide information on the types of factors that might need to be considered when making allocation decisions where technology-based controls on point sources alone are not sufficient to meet water quality standards and a TMDL is thus required.

Assessing alternatives

Each allocation strategy under consideration will need to be tested using the linkage analysis (Section 6) to evaluate the potential effectiveness of the proposed alternative. The analysis should include consideration of the seasonal or annual variability in loadings, particularly where significant contributions are made by precipitation-driven nonpoint sources. As alternative allocation strategies are developed, it might be necessary to reassess the adequacy of the selection of water quality targets and linkages.

Achieving a balance between WLAs and LAs

An appropriate balance should be struck between point source and nonpoint source controls in establishing the formal TMDL components. Finding a balance between WLAs and LAs in a TMDL management unit involves the evaluation of several factors. First, the manager needs to know how the loads causing impairment are apportioned between point and nonpoint sources. Is one source dominating the other? Imposition of controls should reflect the size of the source where possible. For instance, if a pollutant load from a nonpoint source was found to be 80 percent of the total loading to a problem area and a 40 percent overall reduction in loading was needed, necessary load reductions could not be achieved through point source controls alone.

Secondly, the TMDL developer should look at the potential efficacy of controls. What BMP and point source controls are feasible, and how effective will they be? TMDL developers should seek input from the stakeholders on the control preferences and feasibility. Discussion and cooperation among and with stakeholders can result in more successful implementation of the resulting allocation. Time constraints might not allow for an in-depth review in every case, but efforts to gain an understanding of the efficacy of feasible controls will undoubtedly result in more successful TMDL strategies.

Finally, cost-effectiveness should be considered. Since financial resources for controls are limited, emphasis should be placed where possible on allocations that will lead to cost-effective controls.

3. How can TMDLs be translated into controls?

Translate WLAs into NPDES permit requirements

The National Pollutant Discharge Elimination System (NPDES) permit is the mechanism for translating WLAs into enforceable requirements for point sources. The NPDES Program is established in section 402 of the CWA. Under the program, permits are required for the discharge of pollutants from most point source discharges into the waters of the United States (see 40 CFR Part122 for applicability). Although an NPDES permit authorizes a point source facility to discharge, it also subjects the permittee to legally enforceable requirements set forth in the permit. 40 CFR 122.44(d)(1)(vii)(B) requires effluent limits to be consistent with WLAs in an approved TMDL.

One way WLAs are translated into permits is through effluent limitations. Effluent limitations impose restrictions on the quantities, discharge rates, and/or concentrations of specified pollutants in the point source discharge. Effluent limitations reflect either minimum federal or state technology-based guidelines or levels needed to protect water quality, whichever is more stringent. By definition, TMDLs involve WLAs that are more stringent than technology-based limits to protect WQSs and are therefore used to establish appropriate effluent limitations. Effluent limitations may be expressed either as numerical restrictions on pollutant discharges or as best management practices when numerical limitations are infeasible (40 CFR 122.44(k)). NPDES requirements at 40 CFR 122.45(d) require numerical effluent limitations for continuous dischargers to be expressed, unless impracticable, as average weekly and average monthly discharge limitations for POTWs, and as daily maximums and as monthly averages for other dischargers.

Requirements at 40 CFR 122.45(e) provide that non-continuous discharges, such as combined sewer overflows or storm water discharges, must be described and limited in a permit based on consideration of several factors, as appropriate. These factors include the frequency of the discharge, the total mass of the discharge, the maximum rate of discharge of pollutants, and a prohibition or limitation of specified pollutants by mass, concentration, or other measure.

WLAs also can be translated into NPDES permit requirements as part of a general permit, which can be used to address a similar category of discharges, such as storm water. In such instances, the WLA may be allocated to the category of sources subject to the general permit, a subcategory of those sources, or individual sources, based on the permitting authority's assessment of which approach would best control the target pathogens. Depending on the type of discharges covered by the general permit, either numeric effluent limitations or non-numeric controls in lieu of numeric limitations (e.g., best management practices) can be required to achieve the WLA. Numeric effluent limitations are typically applied to relatively continuous discharges or controlled batch discharges. Non-numeric controls are typically used to address non-continuous discharges that tend to be more difficult to model and predict.

There also may be instances where it is advantageous to develop a single WLA that addresses all of the point sources (e.g., POTWs, CSOs, storm sewers) that discharge pathogens within a municipality and allow the permit writer, working with the municipality, to determine how best to allocate the WLA among the relevant point sources. This "municipal integration" approach allows the municipality and permit writer to consider all of the sources of pathogen discharges at the same time and to optimize the allocation between sources based on local treatment system capabilities and control strategies. For example, the EPA 1994 CSO Control Policy encourages POTWs to capture a greater portion of combined sewer overflows for treatment at the municipal wastewater treatment plant (USEPA, 1994d). Other municipalities are considering sewer separation, which will eliminate the contribution of pathogens from CSOs, but increase loadings from municipal storm sewer systems. Municipal integration, which requires a TMDL that encompasses all of the major sources of pathogen discharges within a municipality, provides municipalities with the flexibility to adjust the proportion of flow and loadings between storm water, CSO, and POTW discharge locations to maximize the treatment of sewage and load reductions.

Translate load allocations into implementation plans

Unlike NPDES permits for point sources, there are no corresponding permit requirements for nonpoint sources. Instead, load allocations are addressed, where necessary, through implementation of best management practices (BMPs). In some cases, states have certain mandatory BMP requirements for specific land use activities associated with fecal indicator loads, such as confined animal operations. However, implementation of BMPs usually occurs through voluntary and incentive programs such as government cost sharing. Therefore, when establishing nonpoint source load allocations within a TMDL, the TMDL development documentation should show (1) that there is reasonable assurance that nonpoint source controls will be implemented and maintained or (2) that nonpoint source reductions are demonstrated through an effective monitoring program (USEPA, 1991a, 1999).

Although LAs may be used to target BMP implementation in a watershed, translation of LAs into specific BMP implementation programs can be problematic. One reason for this difficulty is that often many agencies are involved in BMP implementation. Rather than a single oversight agency, as is the case for NPDES permits, BMP implementation can typically include federal, state, and local levels of involvement. Many times the objectives of the varying agencies are different, making coordination difficult.

In addition, it is not always easy to predict the effectiveness of BMPs, particularly in the case of pathogen management. Therefore, it is also difficult to determine the level of effort and resources to focus on BMP implementation to comply with LAs. TMDL strategies that are heavily dependent on loading reductions through LAs should include long-term watershed water quality monitoring programs to evaluate BMP effectiveness and compliance with LAs.

4. How should issues of equity in allocations be addressed?

One issue that arises in distributing assimilative capacity is equity between allocations. Chadderton et al. (1981) provide an examination of a variety of methods to establish WLAs among interacting discharges. The following five methods were reviewed for a situation involving five interacting discharges of biochemical oxygen demand (BOD):

- Equal percent removal or equal percent treatment.
- Equal effluent concentration.
- Equal incremental cost above minimum treatment (normalized on the basis of volumetric flow rate).
- Effluent concentration inversely proportional to pollutant mass inflow rate.
- Modified optimization (i.e., least cost solution that includes the minimum treatment requirements of the technology-based controls).

A comparison of the methods was made based on cost, equity, efficient use of stream assimilative capacity, and sensitivity to fundamental stream quality data. The authors concluded that "equal percent treatment" was preferable in the example studied because of the method's insensitivity to data errors and accepted use by several states. However, although such a method could be used to balance between various point sources or (in some cases) between similar nonpoint sources, it likely would not be feasible for balancing between point and nonpoint sources. The other methods cited by Chadderton et al., or combinations of these methods, might be preferable under different circumstances.

5. How should stakeholders be involved?

In accordance with federal regulations regarding water quality management planning (40 CFR Part 130), TMDLs should be made available for public comment. However, for TMDL strategies to be successful, those parties likely to be effected by the TMDL (the stakeholders) should be involved in the TMDL development process as well. Effective communication is a key element of the public participation process. Stakeholders should be made aware of and engaged in the decisions regarding priority status of a waterbody, the modeling results or data analyses used to establish TMDLs for the waterbody, and the pollutant control strategies resulting from the TMDL (i.e., WLAs and LAs).

EXAMPLE TMDL ALLOCATION

In this simplified example, a river reach receives a steady fecal indicator load from a POTW effluent and an intermittent load from a storm water discharge upstream of the POTW. The relevant state water quality standard is a geometric mean of 400 organisms/100 mL. The storm water discharge has not, however, been sufficiently characterized to make an accurate analysis of the exact statistical distribution of in-stream concentrations. The state is therefore taking the simplified approach of developing a TMDL that is predicted to meet the water quality standard under conditions of mean receiving water flow and event mean flow and fecal indicator concentration from the storm water discharge.

The TMDL is calculated at the mouth of the river. For ease of explication, this example considers only simple mixing with first order die-off in transport from the storm water discharge and the POTW outfall to the river mouth. The receiving water concentration at the river mouth is given by the mass balance equation

$$C_m = \frac{(C_{POTW}\, e^{-kT})\,Q_{POTW} + (C_{SW}\, e^{-kT})\,Q_{SW} + C_R Q_R}{Q_{POTW} + Q_{SW} + Q_R}$$

where

C_m	=	mixed instream concentration;
C_{POTW}	=	concentration in the POTW effluent;
Q_{POTW}	=	flow from the POTW;
C_{SW}	=	concentration in the storm water;
e^{-kT}	=	decay function;
k	=	decay rate;
T	=	travel time between source and river mouth;
Q_{SW}	=	flow from the storm water discharge;
C_R	=	background river concentration; and
Q_R	=	river flow (not including flows from storm water and POTW).

This equation can be rewritten in terms of the load balance as

$$Cm\,(Q_{POTW}+Q_{SW}+Q_R) = C_{POTW}\,e^{-kT}Q_{POTW}+C_{SW}\,e^{-kT}Q_{SW}+C_R Q_R$$

where the left side is the in-stream load and the right side is the sum of loads from sources and background. The loading capacity or TMDL is estimated by replacing the actual C_m with the WQS:

$$TMDL = WQS \cdot \left(Q_{POTW} + Q_{SW} + Q_R\right)$$

The steps for calculating the TMDL and allocations are as follows:

1. Calculate the TMDL that will meet the water quality standard.

2. Calculate the existing loading and MOS.

3. Compare the TMDL with the existing loading plus the MOS (i.e., whether TMDL is exceeded by the loadings plus any reserve and MOS).

4. If the TMDL is exceeded, set allocations by reducing the existing loads to meet the TMDL.

Step 1: Calculate the TMDL that will meet the water quality standard

The data needed to calculate the TMDL (and existing loads in Step 2) are given in Table 7-1.

The TMDL that will meet the water quality standard is calculated as

$$TMDL = \frac{400\ org}{100\ mL}(50+25+100\ ft^3/s) = 700\ \frac{org \cdot ft^3}{s \cdot mL}$$

Table 7-1. Data for calculating the TMDL

Parameter	Symbol	Value
Water Quality Standard	WQS	400 org/100 mL
River flow (mean)	Q_R	100 ft^3/s
POTW flow	Q_{POTW}	50 ft^3/s
SW flow (event mean)	Q_{SW}	25 ft^3/s
Background river concentration	C_R	10 org/100 mL
POTW concentration	C_{POTW}	400 org/100 mL
SW concentration (event mean)	C_{SW}	3000 org/100 mL
Die-off rate	k	0.5 org/day
Travel time from POTW to river mouth	T_{POTW}	0.2 days
Travel time from SW discharge to river mouth	T_{SW}	0.3 days

Step 2: Calculate the existing loading and MOS

If the existing loading to the river together with the MOS (plus any reserve for future growth) exceeds the TMDL calculated in Step 1, a reduction in existing loading is necessary to ensure that the water quality standard will be met. The current loading and MOS that should be compared with the TMDLs are

$$C_{POTW}\,e^{-kT}Q_{POTW}+C_{SW}\,e^{-kT}Q_{SW}+C_R Q_R+MOS+Reserve$$

A reserve is optional, and no reserve is specified in this example. Section 303(d) of the Clean Water Act requires a margin of safety, incorporated either implicitly or explicitly. For this example, the MOS is explicitly defined as 10 percent of the TMDL less the river's background loading.

$$MOS = 10\% \, (TMDL - C_R Q_R)$$

Therefore,

$$MOS = 10\% \text{ x } (700 - 10) = 69 \, \frac{org \cdot ft^3}{s \cdot mL}$$

The existing loading plus MOS to be compared with the TMDL is calculated using

$$\text{Current loading} + MOS =$$

$$172 + 679 + 10 + 69 = 930 \, \frac{org \cdot ft^3}{s \cdot mL}$$

where

$$C_{POTW} e^{-kT} Q_{POTW} = 400 \, \frac{org}{100 \; mL} \text{ x } 0.861 \text{ x } 50 \, \frac{ft^3}{s} = 172 \, \frac{org \cdot ft^3}{s \cdot mL}$$

$$C_{SW} e^{-kT} Q_{SW} = 3000 \, \frac{org}{100 \; mL} \text{ x } 0.905 \text{ x } 25 \, \frac{ft^3}{s} = 679 \, \frac{org \cdot ft^3}{s \cdot mL}$$

$$C_U Q_U = 10 \, \frac{org}{100 \; mL} \text{ x } 100 \, \frac{ft^3}{s} = 10 \, \frac{org \cdot ft^3}{s \cdot mL}$$

Step 3: Compare the TMDL with the existing loading plus the MOS

At the specified hydrologic conditions, the existing loading and MOS exceed the TMDL, so it is necessary to reduce the existing loadings to set the allocations.

Step 4: Set allocations

Numerous wasteload allocation methods can be used for calculating necessary reductions in loads to meet the TMDL. The method in this example is to first determine the portion of the TMDL available for allocation to known sources (*Alloc*),

$$Alloc = TMDL - Background - MOS$$

and then to reduce the loads to match this allocatable fraction. The reduced loads are calculated by multiplying *Alloc* by the fraction allocated to each individual source

$$WLA_i \text{ (or } LA_i) = Alloc \cdot f_i$$

where

WLA_i	=	wasteload allocation for point source *i*;
LA_i	=	load allocation for nonpoint source *i*;
Alloc	=	portion of the TMDL allocatable to sources; and
f_i	=	fraction of the allocatable load assigned to source *i*.

The allocation fraction assigns reductions proportional to existing load (other allocation schemes are, of course, possible). It is used to calculate reduced loading for the individual sources that taken together will meet the TMDL, while maintaining the existing percent contribution of the individual source loads

$$f_i = \frac{L_i}{\sum L_i}$$

where:

L_i	=	existing load from source *i* and
ΣL_i	=	total existing loading from all significant identified sources.

Using the existing loads for each source from Step 2, the allocatable portion of the TMDL is calculated as

$$Alloc = TMDL - Background - MOS$$

Therefore,

$$Alloc = 700 - 10 - 69 = 621 \, \frac{org \cdot ft^3}{s \cdot mL}$$

The allocation fractions, f_i, are calculated as

$$f_{POTW} = \frac{L_{POTW}}{L_{POTW} + L_{SW}} = \frac{172}{851} = 0.202$$

and

$$f_{SW} = \frac{L_{SW}}{L_{POTW} + L_{SW}} = \frac{679}{879} = 0.798$$

Allocations are then assigned using

$$WLA_{POTW} = 621 \cdot 0.202 = 125 \ \frac{org \cdot ft^3}{s \cdot mL}$$

$$WLA_{SW} = 621 \cdot 0.798 = 496 \ \frac{org \cdot ft^3}{s \cdot mL}$$

while the LA for background is, as noted above, 10 org-ft^3/s-mL.

Totaling the reduced loads according to the allocations and including background and MOS results in the following:

$$WLA_{POTW} + WLS_{SW} + MOS + LA_{Background} = TMDL$$

or

$$142 + 480 + 69 + 9 = 700 \ \frac{org \cdot ft^3}{s \cdot mL}$$

or

$$3.54x10^6 + 1.41x10^7 + 1.95x10^6 + 2.83x10^5 = 1.99x10^7 \ \frac{org}{s}$$

$$= 1.72x10^{12} \ \frac{org}{day}$$

Thus, the allocations meet the TMDL under the specified conditions. Because of the simplified nature of the analysis, however, continued monitoring would be needed to ensure that the water quality standard is maintained.

Summary

The allocation step translates the TMDL into allowable loads, which are distributed among the various sources, and also accounts for a margin of safety and seasonal variation. Allocations are required for both point sources (WLAs) and nonpoint sources (LAs) and must include either an implicit or explicit margin of safety (MOS). Point source WLAs can be translated into NPDES permit requirements; nonpoint source LAs can be translated into implementation plans. The TMDL implementation plan for point and nonpoint sources may be submitted with the TMDL. However, the plan is not a component of the actual TMDL and is not approved or disapproved by EPA. Because the allocations will involve issues such as equity, economics, and politics, it is important that the administering agency involve stakeholders throughout development of the TMDL.

RECOMMENDATIONS FOR ALLOCATIONS

- Identify the method of incorporating the margin of safety (i.e., implicitly or explicitly).
- Reflect the relative size and magnitude of sources, where possible, and represent an appropriate and feasible balance between and among WLAs and LAs.
- Include adequate documentation with allocations to provide reasonable assurance that water quality standards will be attained and TMDL will be implemented.
- Involve affected stakeholders in the development of allocations.

RECOMMENDED READING

(Note that a full list of references is included at the end of this document.)

Chadderton, R.A., A.C. Miller, and A.J. McDonnell. 1981. Analysis of waste load allocation procedures. *Water Resources Bulletin* 17(5):760-766.

Freedman, P.K., and J.K. Marr. 1990. Receiving-water Impacts. In *Control and Treatment of Combined Sewer Overflows*. pp. 79-117. Van Nostrand Reinhold, New York.

Thomann, R.V., and J.A. Mueller. 1987. *Principles of Surface Water Quality Modeling and Control.* Harper & Row, New York, NY.

USEPA. 1991a. *Guidance for Water Quality-based Decisions: The TMDL Process.* EPA 440/4-91-001. U. S. Environmental Protection Agency, Assessment and Watershed Protection Division, Washington, DC.

USEPA. 1991b. *Technical Support Document for Water Quality-based Toxics Control.* EPA/505/2-90-

001. U.S. Environmental Protection Agency, Office of Water, Washington, DC.

USEPA. 1993a. *Guidance Specifying Management Measures for Sources of Nonpoint Pollution in Coastal Waters*. EPA 840-B-92-002. U.S. Environmental Protection Agency, Office of Water, Washington, DC.

USEPA. 1995a. *Watershed Protection: A Project Focus*. EPA 841-R-95-003. U.S. Environmental Protection Agency, Office of Water, Washington, DC.

USEPA. 1999. *Draft Guidance for Water Quality-based Decisions: The TMDL Process*. 2nd ed. EPA 841-D-99-001. U.S. Environmental Protection Agency, Washington, DC.

Follow-up Monitoring and Evaluation

Objective: Define a monitoring and evaluation plan to validate TMDL elements, assess the adequacy of control actions to implement the TMDL, and provide a basis for reviewing and revising TMDL elements or control actions in the future.

Procedure: Identify the key questions a monitoring plan needs to address and evaluate monitoring options and the feasibility of implementing a monitoring program. Describe a specific monitoring plan, including the timing and location of monitoring activities, parties responsible for conducting monitoring, and quality assurance/quality control procedures. Provide the schedule for reviewing monitoring results to consider the need for TMDL or action plan revisions, and discuss the adaptive management approach to be taken. The monitoring component of a TMDL results in a description of monitoring and adaptive management plan objectives, methods, schedules, and responsible parties.

Key Questions to Consider for Follow-up Monitoring and Evaluation

1. What factors influence the monitoring plan design?
2. What is an appropriate monitoring plan?
3. What is an appropriate adaptive management plan, including review and revision schedule?
4. What is an adequate description of the monitoring plan for the TMDL document?

OVERVIEW

TMDL submittals should include a monitoring plan to determine whether the TMDL has attained water quality standards and to support any revisions to the TMDL that might be required. Follow-up monitoring is recommended for all TMDLs, given the uncertainties inherent in TMDL development (USEPA 1991a; 1997b). The rigor of the monitoring plan should depend on the confidence in the TMDL analysis: a more rigorous monitoring plan should be included for TMDLs with more uncertainty and where the public health, environmental, or economic consequences of the decisions are harshest (*e.g.*, in protecting public drinking water supplies.) This section discusses key factors to consider in developing the monitoring plan and suggests additional sources of guidance on monitoring plan development.

Models often can prove useful in evaluating the results of monitoring. Because weather and other watershed process drivers usually are not identical before and after implementation, it is difficult to compare monitoring data results. The monitoring must consider that situation. If models are calibrated to conditions before and after implementation, they then can be run for the post-implementation period assuming implementation practices are not applied. This approach can facilitate the evaluation of the relative effectiveness of different implementation approaches and the adequacy of different TMDL components.

Compliance monitoring by public water systems may also be useful where public water supply protection is at issue. As noted earlier, drinking water treatment is designed to remove a proportion, but not all, of the pathogen contamination in the influent. Therefore, higher pathogen loadings in the waterbody translate into higher pathogen contamination levels in the treated water and a greater public health risk. If a public water system experiences repeated exceedances of the monitoring trigger levels for pathogens (*e.g.*, turbidity or total coliform), one or more of the TMDL control measures may need refining.

KEY QUESTIONS TO CONSIDER FOR FOLLOW-UP MONITORING AND EVALUATION

1. What factors influence the monitoring plan design?

Key factors to consider in developing the TMDL monitoring plan include the following:

Need to evaluate specific TMDL elements

TMDL problem identification, indicators, numeric targets, pollutant load estimates, and allocations might need to be reevaluated to determine if they are accurate and effective. The monitoring plan should define specific questions to be answered about these elements through the collection of monitoring information. Potential questions include the following:

- Are the selected indicators capable of detecting designated or existing use impacts of concern and responses to control actions?
- Have baseline or background conditions been adequately characterized?
- Are the numeric targets set at levels that reasonably represent the appropriate desired conditions for designated or existing uses of concern?
- Have all important pollutant sources been identified?
- Have pollutant sources been accurately estimated?
- Has the linkage between pollutant sources and impacts on the waterbody been accurately characterized?
- Have other watershed processes (e.g., hydrology) that affect pathogen loads or affect designated or existing uses been accurately characterized?
- Where reference sites were used to help determine TMDL targets and load reduction needs, were reference site conditions accurately characterized?
- Were models or methods used for the TMDL accurately calibrated, validated, and verified?

Not all questions are appropriate for all TMDL monitoring plans because the degree of uncertainty concerning different TMDL elements will vary from case to case.

Need to evaluate implementation actions

It is often important to determine whether actions identified in the implementation plan were actually carried out (implementation monitoring) and whether those actions were effective in attaining TMDL allocations (effectiveness monitoring). Specific questions to be answered concerning implementation actions should be articulated as part of the monitoring plan.

Stakeholders' goals for monitoring efforts

Watershed stakeholders often participate in follow-up monitoring, and their interests should be considered in devising monitoring plans.

Existing monitoring activities, resources, and capabilities

Analysts should identify existing and planned monitoring activities to coordinate TMDL monitoring needs with other planned efforts, particularly where a long-term monitoring program is envisioned, the study area is large, or water quality agency monitoring resources are limited. Staff capabilities and training should also be considered to ensure that monitoring plans are feasible.

Practical constraints to monitoring

Monitoring options are often limited by practical constraints such as problems with access to monitoring sites or concerns about the indirect impacts of monitoring on habitat. Other factors that influence the design of monitoring plans and different types of monitoring of interest for TMDLs are discussed in detail in MacDonald et al. (1991).

Types of monitoring

Several types of monitoring may be considered in developing the monitoring plan (modified from McDonald et al., 1991):

- *Baseline monitoring*. Baseline monitoring characterizes existing conditions and provides a basis for future comparisons. This type of monitoring is not always necessary for the monitoring plan. Usually, some baseline data that were considered during TMDL development already exist.
- *Implementation monitoring*. This type of monitoring would ensure that identified management actions (such as specific BMPs or resource restoration or enhancement projects) are undertaken. This information would also be analyzed as one of the factors that influences the conclusions from the trend monitoring.

Characteristics of an Effective Monitoring Plan

- Quantifiable approach. Results must be discernible over time, to allow comparison to previous or reference conditions.
- Appropriate in scale and application, and relevant to designated or existing uses and the TMDL methods.
- Adequately precise, reproducible by independent investigators, and consistent with scientific understanding of the problems and solutions.
- Able to distinguish among many different factors/sources (e.g., pasture/feedlot runoff, urban runoff, septic systems, wildlife).
- Versatile. Generally looks at the problem from a number of different perspectives.
- Understandable to the public and supported by stakeholders.
- Feasible and cost-effective.
- Anticipates potential future conditions and climatic influences.
- Minimizes disruptions to the designated or existing uses while collecting data.
- Facilitates reaching and sustaining conditions that support the designated or existing use.

- *Project or effectiveness monitoring*. Specific projects undertaken in the context of the TMDL, or separate from the TMDL, but potentially affecting water quality conditions for the watershed area under consideration, should be monitored to determine both their immediate effects and the effects on the water quality downstream of the project.

- *Trend monitoring*. This type of monitoring assesses the effectiveness of management actions and the changes in conditions over time relative to the baseline and identified target values. Trend monitoring is the primary type of follow-up monitoring, assuming the other elements of the TMDL are appropriately developed. It addresses the changing conditions in the waterbody that result from TMDL-specific activities, as well as other land management activities, over time. Trend monitoring is the most critical component of the monitoring program since it also documents progress toward achieving the desired water quality conditions.

- *Validation monitoring*. This type of monitoring is used to reevaluate the selection of indicators, numeric targets, and/or source analysis methods.

2. What is an appropriate monitoring plan?

Identify monitoring goals

Depending on the level of precision in the TMDL analysis, the first step in developing an appropriate monitoring and adaptive management plan is to clearly identify the goals of the monitoring program. It may be possible to accomplish several of these monitoring goals simultaneously. For example, the primary need in most TMDLs is to document progress toward achieving the numeric targets. During this process, the additional information collected might lead to a better understanding of the processes, suggesting a revision to the source analysis that would better pinpoint the pathogen problem and lead to faster attainment of water quality improvements, or it may be that a particular restoration/enhancement project did not produce the desired effects and some changes to it should be undertaken.

Develop and articulate the hypothesis and experimental design

Address the relationships between the monitoring plan and the TMDL's numeric targets, source analysis, linkages, and allocations, as well as the implementation plan. Articulate specific questions to be answered in the form of monitoring hypotheses, and explain how the monitoring program will answer those questions. Explain any assumptions being made. Explain how the monitoring plan will address both episodic events and continuous effects, and discuss the likely effects of episodic events. The design can be delineated by source type, by geographical area, or by ownership parcel.

Discuss procedural details

Describe the monitoring methods to be used and provide rationale for selection of these methods. Define monitoring locations and frequencies, and specify who will be responsible for conducting the monitoring (if known).

Develop an appropriate quality assurance project plan

Detail sampling methods, selection of sites, and analysis methods consistent with accepted quality assurance and quality control practices. Have the monitoring plan peer reviewed if possible. (For more information see USEPA, 1994b, 1994c.)

3. What is an appropriate adaptive management plan, including review and revision schedule?

The plan should contain a section addressing the adaptive management component. This section should discuss when and how the TMDL will be reviewed. If possible, the plan should describe criteria that will guide TMDL review and revision. For example, the plan could identify expected levels of progress toward meeting TMDL numeric targets at the time of the initial review, stated as interim numeric targets or interim load reduction expectations. In addition, the plan could identify "red flag" thresholds for key indicators that would signal fundamental threats to designated or existing uses and perhaps trigger a more in-depth review of the components of the TMDL and implementation plan.

The adaptive management component need not schedule every TMDL review that will ever be needed; it should be adequate to indicate the estimated frequency of review and identify a specific date for the initial review. It would be difficult to reliably forecast how often TMDL reviews will be needed, especially where problems will take several years (or more) to solve.

4. What is an adequate description of the monitoring plan for the TMDL document?

The monitoring and adaptive management plan is a required component of TMDLs developed under the phased approach (USEPA, 1991a). The plan should incorporate each of the components discussed above along with adequate rationale for the selected monitoring and adaptive management approach. If it is infeasible to develop the monitoring plan in detail at the time of TMDL adoption, it may be adequate to identify the basic monitoring goals, review the time frame, and identify responsible parties while committing to develop the full monitoring plan in the near future. The plan should clearly indicate the monitoring goals and hypotheses, the parameters to be monitored, the locations and frequency of monitoring, the monitoring methods to be used, the schedule for review and potential revision, and the parties responsible for implementing the plan.

RECOMMENDATIONS FOR FOLLOW-UP MONITORING AND EVALUATION

- Clearly identify the goals of the monitoring program.
- Define specific questions to be answered concerning the evaluation of individual TMDL elements.
- If possible, coordinate with other existing or planned monitoring activities.
- Determine which type(s) of monitoring (e.g., implementation, trend, etc.) is appropriate for accomplishing the desired goals.
- Develop an appropriate quality assurance plan; follow-up monitoring should be designed to yield defensible data that can support future analysis.

RECOMMENDED READING

(Note that a full list of references is included at the end of this document.)

MacDonald, L., A.W. Smart, and R.C. Wissmar. 1991. *Monitoring Guidelines to Evaluate Effects of Forestry Activities on Streams in the Pacific Northwest and Alaska*. EPA 910/9-91-001. U.S. Environmental Protection Agency, Region 10, Nonpoint Source Section, Seattle, WA.

USEPA. 1992b. *Monitoring Guidance for the National Estuary Program*. EPA 842 B-92-004. U.S. Environmental Protection Agency, Washington, DC.

USEPA. 1996c. *Nonpoint Source Monitoring and Evaluation Guide*. Draft final, November 1996. U.S. Environmental Protection Agency, Office of Water, Washington, DC.

Assembling the TMDL

Objective: Clearly identify components of a TMDL submittal in order to support adequate public participation and to facilitate TMDL review and approval.

Procedure: Compile all pertinent information used to develop the TMDL and prepare the final submittal. The final submittal should document all major assumptions and analyses.

OVERVIEW

It is important to clearly identify the "pieces" of the TMDL submittal and show how they fit together to provide a coherent planning tool that can lead to attainment of water quality standards for pathogen-related water quality impairments. Where TMDLs are derived from other analyses or reports, it is helpful to develop a separate document or chapter that ties together the TMDL components and shows where background information on the derivation of each component can be found.

RECOMMENDATIONS REGARDING CONTENT OF SUBMITTALS

Section 303(d) of the CWA and EPA's implementing regulations provide that a TMDL consists of the sum of WLAs for future and existing point sources and LAs for future and existing nonpoint sources and natural background. These loads are established at levels necessary to implement applicable water quality standards with consideration of seasonal variation and a margin of safety. Experience indicates, however, that information in addition to the statutory and regulatory requirements may be needed to support adequate public participation and to facilitate EPA review and approval. As partners in the TMDL development process, it is in the best interest of the state and EPA to work together to determine how much supporting information is needed in the TMDL submittal.

Recommended minimum submittal information

The following list of TMDL submittal elements provides a suggested outline for TMDL submittals:

1. Submittal Letter
   - Each TMDL submitted to EPA should be accompanied by a submittal letter stating that the submittal is a draft or final TMDL submitted under § 303(d) of the CWA for EPA review and approval.

2. Problem Statement
   - Waterbody name and location.
   - A map is especially useful if information displayed indicates the area covered by the TMDL (e.g., watershed boundary or upper and lower bounds on the receiving stream segment) and the location of sources.
   - Waterbody § 303(d) list status (including pollutant covered by the TMDL and priority ranking).
   - Watershed description (e.g., predominant land cover or land use, geology, and hydrology).

3. Applicable Water Quality Standards and Water Quality Numeric Targets
   - Description of applicable water quality standards, including designated use(s) affected by the pollutant of concern, numeric or narrative criteria, and the antidegradation policy.
   - If the TMDL is based on a target other than a numeric water quality standard, describe the process used to derive the target.

4. Pollutant Assessment
   - Source inventory, including magnitude and location of
     - Background
     - Point sources
     - Nonpoint sources
   - Supporting documentation for the analysis of pollutant loads from each source.

5. Linkage Analysis
   - Rationale for the analytical method used to establish the cause-and-effect relationship between the numeric target and the identified pollutant sources.
   - Supporting documentation for the analysis (e.g., basis for assumptions, strengths and weaknesses

in the analytical process, results from water quality modeling).

6. TMDL and Allocations
   - Total Maximum Daily Load (TMDL)[1]
     - The TMDL is expressed as the sum of the WLAs, the LAs, and the MOS (if an explicit MOS is included).
     - If the TMDL is expressed in terms other than mass per time, explain the selection of the other appropriate measure.
   - Wasteload Allocations (WLAs)[2]
     - Loads allocated to existing and future point sources.
     - An explanation of any WLAs based on the assumption that loads from a nonpoint source will be reduced.
     - If no point sources are present, list the WLA as zero.
   - Load Allocations (LAs)[2]
     - Loads allocated to existing and future nonpoint sources.
     - Loads allocated to natural background (where possible to separate from nonpoint sources).
     - If there are no nonpoint sources and/or natural background, the LA should be listed as zero.
   - Seasonal Variation[1]
     - Description of the method chosen to take into account seasonal and interannual variation.
   - Margin of Safety[1]
     - An implicit MOS is accounted for through conservative assumptions in the analysis. To justify this type of margin of safety, an explanation of the conservative assumptions used is needed.
     - An explicit MOS is incorporated by setting aside a portion of the TMDL as the MOS.
   - Critical Conditions[2]
     - Critical conditions associated with flow, loading, designated use impacts, and other water quality factors.

7. Follow-Up Monitoring Plan
   - Recommended component for TMDLs.

8. Public Participation[2]
   - Description of public participation process used.
   - Summary of the significant comments received and the responses to those comments.

9. Implementation Plan
   - Implementation plans are needed before TMDL approval if they are necessary to provide reasonable assurance that the load allocations contained in the TMDL will be achieved.

Supplementary TMDL submittal information

In addition to the information described above, TMDL submittals can be improved by preparing supplemental information, including a TMDL summary memorandum, a TMDL executive summary, a TMDL technical report, and an administrative record. The effort required to develop these documents should be minimal because they are largely a repackaging of information contained in the TMDL submittal. For example, the TMDL executive summary would be prepared for the TMDL technical report but would also be ideal for press releases or distribution to the public.

The *TMDL summary memorandum* provides an overview of all the essential regulatory elements of a TMDL submittal. This overview can assist in regulatory and legal review. The summary memo should include the following information:

- Name, size, and location of waterbody
- Pollutant of concern
- Primary pollutant source(s)
- Applicable water quality standards
- Major data and information sources
- Linkage analysis and load capacity (TMDL establishment)
- WLA, LA, MOS, critical condition, seasonality, background concentrations
- Implementation
- Reasonable assurance
- Follow-up monitoring
- Public participation

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[1] Required by statute.

[2] Required by regulation.

The *TMDL executive summary* provides an overview of the TMDL, the conclusions and implications, the analyses, and the background. This document is useful for public information, news releases, and public hearing announcements.

The *TMDL technical report* provides a compilation of the information sources, technical analyses, assumptions, and conclusions. This document provides a summary of the technical basis and rationale used in deriving the TMDL. A sample report outline might include the following sections:

1. Executive Summary
2. Introduction
3. TMDL Indicators and Numeric Targets
4. Water Quality Assessment
5. Source Assessment
6. Linking the Sources to the Indicators and Targets
7. Allocation
8. Implementation
9. Monitoring
10. References

The *administrative record* provides the technical backup, sources of information, calculations, and analyses used in deriving the TMDL. After-the-fact explanations or justifications of EPA's decisions are generally not permitted. A typical administrative record might include the following:

- Spreadsheets
- Modeling software, input/output files
  - Description of the methodology/models used, and a description of the data used for the models.
- References
  - List or index of all documents relied upon by the state or EPA in making a decision.
- Reports
  - Any EPA documents (i.e., national/regional guidance, interpretations, protocols, technical documents relied upon in making a decision).
  - Comments/correspondence from outside parties and EPA's or state's responses, including copies of public notice seeking comment, and final decision document.
- Communication
  - Documentation of communication between EPA and the state or EPA and other federal agencies regarding the TMDL.
- Paper calculations
- Maps (working copies)

Public participation

Public participation is a requirement of the TMDL process and is vital to a TMDL's success. EPA believes that stakeholders can contribute much more than their comments on a specific TMDL during the public review process. Given the opportunity, stakeholders can contribute credible, useful data and information about an impaired or threatened waterbody. They may also be able to raise funds for monitoring or to implement a specific control action and/or management measure.

More importantly, stakeholders can offer insights about their community that may ensure the success of one TMDL allocation strategy over an alternative, as well as the success of follow-up monitoring and evaluation activities. Stakeholders possess knowledge about a community's priorities, how decisions are made locally, and how different residents of a watershed interact with one another. A thorough understanding of the social, political, and economic issues of a watershed is as critical to successful TMDL development as an understanding of the technical issues. States, territories, and authorized tribes can create a sense of ownership among watershed residents and "discover" innovative TMDL strategies through a properly managed public participation process.

Each state, territory, and authorized tribe is required to establish and maintain a continuing planning process (CPP) as described in § 303(e) of the Clean Water Act. A CPP contains, among other items, a description of the process used to identify waters needing water quality-based controls, a priority ranking of such waters, the process for developing TMDLs, and a description of the process used to receive public review of each TMDL. EPA encourages states, territories, and authorized tribes to use their CPP as the basis for establishing a process for public participation, involvement, and in many cases leadership in TMDL establishment. On a watershed level, the continuing planning process allows programs to combine or leverage resources for public outreach and involvement, monitoring and assessment, development of management strategies, and implementation.

RECOMMENDED READING

(Note that a full list of references is included at the end of this document.)

USEPA. 1991a. *Guidance for Water Quality-based Decisions: The TMDL Process.* EPA 440/4-91-001. U.S. Environmental Protection Agency, Assessment and Watershed Protection Division, Washington, DC.

USEPA. 1999. *Guidance for Water Quality-based Decisions: The TMDL Process.* 2nd ed. EPA 841-D-99-001. U.S. Environmental Protection Agency, Assessment and Watershed Protection Division, Washington, DC.

APPENDIX A: How Pathogen Indicators Are Measured

The methods for measuring densities of the bacterial indicators on which water quality standards are based have evolved over the years; standard methods are now available for total and fecal coliforms, enterococci, and *E. coli*. Specially equipped microbiological laboratories and highly trained technicians are usually required to conduct these tests, and appropriate quality assurance and quality control procedures must be followed to reduce uncertainties in the estimates of the pathogens. Basic methods are presented in the 19th edition of *Standard Methods for the Examination of Water and Wastewater* (APHA, 1995). Newer and improved methods are now being developed and tested for some groups of pathogens, especially viruses and protozoans. Few techniques are able to distinguish between human and animal wastes, although the ability to do this has been explored in some studies as a means of tracing the sources of pathogens. Table A-1 summarizes some commonly used measurement methods for pathogens and indicator bacteria in surface water samples, as well as some of the newer measurement methods under consideration, and they are described briefly in this section.

Different types of gram-negative bacteria can be distinguished based on their ability to survive, grow, and reproduce in the presence of a particular organic compound, known as a substrate. Sometimes they are distinguished by their ability to produce a particular metabolic by-product, such as methane gas, their ability to change the color of a compound, or to produce fluorescence. Bacteria are also distinguished based on the color, shape, or other characteristics of the colony that is formed when they are grown on a particular organic substrate. In general, two procedures, the multiple-tube fermentation technique and the membrane filter technique, are commonly used to identify fecal bacteria.

The multiple-tube fermentation technique was developed first. A set of tubes containing enriched broth are inoculated with different amounts of the water sample and incubated at a specific temperature for a predetermined period of time. If gas is produced in the tube, a sample of the bacteria in the broth is transferred to one or more additional media to confirm the presence of fecal coliform bacteria. Additional biochemical tests can be performed to identify the bacteria to genus and species or higher, in order to verify that the bacteria found are coliforms (Pepper et al., 1995). The number of tubes producing gas are converted to express the results of the test as the Most-Probable-Number (MPN) per 100 mL water, a statistical estimation of the number of coliform bacteria that would give the results shown by the laboratory examination. This is a statistical probability number and is not an actual enumeration. This method may give higher results because of the built-in 23 percent positive bias.

The membrane filter technique was developed later to detect and quantify the bacteria found in a water sample (Pepper et al., 1995; USEPA, 1986). A measured amount of sample is filtered through a membrane with a pore size of 0.45 μm. The bacteria are retained on the membrane and the filter is placed on the surface of a selective agar medium and incubated at a specific temperature for a specified period of time. Following incubation, the colonies formed by the growth of the bacterial cells are counted under a microscope using low magnification. The membrane filter technique thus provides an estimate of the number of coliform bacteria that form colonies when cultured (colony-forming units or CFU per 100 mL). Since some of the colonies could be formed from more than one bacterium, the count is considered to be an estimate.

USEPA currently recommends the membrane filtration procedure because it is faster and more precise than the MPN technique; however, it is more complex and requires greater interpretive expertise by the analyst (NRDC, 1996). Parallel tests using both procedures should be performed to demonstrate applicability and comparability if they have not been used before (Grandi et al., 1989). Waters with high turbidity or high noncoliform (background) bacterial levels can interfere with the membrane filtration procedure by clogging the filter or suppressing coliform growth respectively. *E coli* and fecal streptococci can also be detected by the membrane filter procedure. (A new video on the improved enumeration methods for *E. coli* and enterococci is available from USEPA's Office of Water, Standards and Applied Science Division, Water Quality Standards Branch).

Another procedure, the Autoanalysis Colilert test, was developed to detect total coliforms and *E. coli* in water samples. It can be performed within 18-24 hours, and a modification allows this test to be used with highly

Table A-1. Potential measurement endpoints for some pathogens and indicator bacteria

Group	Indicator Organisms	Method (Reference)
Viruses	F1 Coliphage	9211, Coliphage Detection (Proposed) (APHA, 1995)
	MS2 Bacteriophage	Adams (1959)
	Poliovirus type 1 strain LSc2ab	Smith and Gerba (1982)
	Enteroviruses	ICR method (USEPA, 1996d)
Coliform Bacteria	Total Coliform	9132, Membrane Filtration Technique; 9131, Multiple Tube Fermentation Technique (Chapter 5 in USEPA, 1984b); 9221, Total Coliform Fermentation Technique; 9222, Total Coliform Membrane Filter Procedure; 9223, Chromogenic Substrate Coliform Test (APHA, 1995)
	Fecal Coliform	(USEPA, 1978) 9221, Fecal Coliform Procedure and 9222, Fecal Coliform Membrane Filter Procedure (APHA, 1995)
	Escherichia coli	1103.1 (USEPA, 1985) 9213, Tests for *E. coli* and 9223, Chromogenic Substrate Coliform Test (APHA, 1995)
	Pseudomonas aeruginosa	9213, Membrane Filter Technique for *Psuedomonas aeruginosa* (APHA, 1995)
	Klebsiella spp.	9222, *Klebsiella* Membrane Filter Procedure (APHA, 1995)
Enterococci Bacteria	*Enterococcus faecalis* *Enterococcus faecium*	Levin et al. (1975) (USEPA, 1978) 1106.1 (USEPA, 1985) 9230, Multiple Tube Fermentation or Membrane Filter Techniques (APHA, 1995) EPA method 1600
	Staphylococcus aureus	9213, Test for *Staphylococcus aureus* (APHA, 1995)
Protozoa	*Cryptosporidium* spp. *Giardia* spp.	9711, Immunofluorescence Method for *Giardia* and *Cryptosporidium* spp. (Proposed) (APHA, 1995) EPA method 1623

turbid samples (Bitton et al., 1995). In this test, *E. coli* are those coliform bacteria which possess the enzyme β-glucuronidase and are capable of cleaving the fluorogenic substrate, 4-methylumbelliferyl-β-D-glucuronide (MUG), with the corresponding release of the fluorogen. This same principle is used in the detection of *E. coli* in EC-MUG medium, used in the MPN method and incubated at 44.5 °C for 24 hours.

Other indicators and methods are under development. Most of the indicators are indirect and warn of the possible presence of fecal pathogens, but not necessarily from humans and potentially from several sources. Some direct indicators, such as the bacteria *Shigella* or *Staphylococcus aureus* or poliovirus, are highly specific for humans but are not usually measured. Other species of bacteria can also cause disease in humans or are more likely to be found in human feces; however, their detection requires special techniques (e.g., gram-positive spore-forming *Clostridium perfringens* and *Campylobacter*). The usual biochemical tests to distinguish *Campylobacter* have been considered unsatisfactory, but other methods have been developed that permit more rapid identification, including agglutination assays, DNA hybridization tests, and polymerase chain reactions (PCR) (Koenraad et al., 1997). These types of methods are also needed to detect bacteria that transform into a viable, but nonculturable stage under unfavorable environmental conditions (Rollins and Colwell, 1986). "Stressed organisms" is the term used to refer to indicator bacteria that become injured in waters and wastewaters. These organisms are unable to grow or reproduce to form colonies under the usual culture conditions (i.e., they are viable, but nonculturable) because of structural or metabolic damage from a variety of factors, including partial or inadequate disinfection, heavy metals, ultraviolet light, and extremes of pH and temperature. False negative microbiological test results, in which some of the indicator bacteria present are not detected and standards are not exceeded, could suggest that a waterbody is safe for its designated use when, in fact, it is not. Injured organisms may retain the potential for virulence and may recover after being ingested (reviewed in APHA, 1995). A method to enhance recovery when culturing viable, but nonculturable organisms is provided in *Standard Methods for the Examination of Water and Wastewater* (APHA, 1995).

Methods are also available or under development to detect enteric viruses and viruses that infect fecal bacteria (coliphages). These methods usually require cell cultures, and specific cells (e.g., bacteria, liver, kidney, nerve, gastrointestinal epithelium) need to be cultured to detect specific viruses (APHA, 1995). Immunofluorescent antibody procedures can also be used for identifying specific viruses. Alderisio et al. (1996) described a method for identifying the four serogroups of male-specific, or F+, RNA coliphages (viruses that infect fecal coliform bacteria). Two of these serogroups are known only from humans and the other two infect fecal coliform from nonhuman sources, which might help in developing an appropriate TMDL allocation. However, none of these methods are associated with water quality standards.

Methods for detecting the encysted parasites *Cryptosporidium* spp. and *Giardia* spp. have been developed (Pepper et al., 1995; USEPA, 1996d) and are being used to evaluate densities of these pathogens in surface waters and drinking water. However, *Giardia* spp. cysts and *Cryptosporidium* spp. oocysts are difficult to isolate from surface water samples, and detection requires the use of special microscopy equipment, complex staining procedures, and a trained analyst. A fluorescent antibody that specifically binds to the cysts and oocysts is used to assist in the enumeration of the parasites. Water samples are obtained by pumping 350 to 1500 L of water through polypropylene yarn-wound cartridge filters using a gasoline-powered pump (Rose et al., 1988). After sample collection, the filters are washed with a solution to rinse the particles off the filter yarn or are cut into pieces and the fibers teased apart and homogenized with this solution. The sample is processed through several steps to separate the cysts and oocysts, which are collected on a cellulose nitrate or a cellulose acetate membrane filter, and the antibody is applied to the samples on the filters. The filters are then mounted on slides and examined using epifluorescence microscopy. In addition to specific immunofluoresence, size, shape, and internal morphology are also examined using phase contrast or differential interference contrast microscopy to distinguish these protozoans. The volume of water sampled, number of cysts and oocysts present, and water turbidity are the major factors influencing the identification of these parasites. Because recoveries from surface water samples have often been low, resulting in the underestimation of parasite densities, other procedures are being evaluated (Newman, 1995). In addition, the antibodies currently used cannot distinguish species of the parasites and thus species that are not pathogenic to humans are included in the counts. Additional work is under way to develop molecular probes for the differentiation of species of *Cryptosporidium* and *Giardia*; to improve the immunologic methods used to detect, identify, and enumerate these organisms; and to determine the percentage of oocysts and cysts in any sample that are viable and infectious to humans (reviewed in Adam, 1991; Mahbubani et al., 1991, 1992; USEPA, 1993b; Webster et al., 1993).

APPENDIX B: Case Study

Muddy Creek, Virginia, TMDL

TMDL Summary: Muddy Creek, Virginia [1]

Waterbody Type:	Stream
Pollutant:	Fecal coliform bacteria
Designated Uses:	Recreational uses; the propagation and growth of a balanced, indigenous population of aquatic life, including game fish, which might reasonably be expected to inhabit them; wildlife; and the production of edible and marketable natural resources (e.g., fish and shellfish) (9VAC 25-260-10).
Size of Waterbody:	10.36 miles
Size of Watershed:	20,025 acres
Water Quality Standards:	Fecal Coliform: Maximum shall not exceed 1,000 fecal coliform/100mL at any time or a geometric mean criterion of 200 fecal coliform/100 mL based on two or more samples collected within a 30-day period
Indicators:	Same as water quality standards
Analytical Approach:	USEPA's BASINS modeling system

INTRODUCTION

The Virginia Department of Environmental Quality has identified Muddy Creek as being impaired by fecal coliform bacteria for a length of 10.36 miles, as reported on the 1998 303(d) list of water quality limited waters. Muddy Creek is prioritized as "high" on the list.

The Muddy Creek watershed is located in Rockingham County, Virginia, approximately 10 miles to the west-northwest of Harrisonburg, Virginia. Muddy Creek flows south to its connection with the Dry River, which discharges to the North River approximately 2.25 miles farther to the south. The North River flows to the South Fork of the Shenandoah River, a tributary of the Potomac River, which eventually discharges into the Chesapeake Bay. The land area of the Muddy Creek watershed is approximately 20,025 acres, and forest and agriculture are the primary land uses. Rockingham County is the largest agricultural county in Virginia for dairy and poultry production. A majority of the agricultural land is located in the central and the eastern portions of the watershed; the forested areas are generally located in the western portion.

The TMDL developed for Muddy Creek illustrates the steps that can be taken to address a waterbody impaired by elevated levels of fecal coliform bacteria. The plan is consistent with a phased-approach TMDL: estimates are made of needed reductions of pollutant loads, load-reduction controls are implemented, and water quality is monitored to determine plan effectiveness. Flexibility is built into the plan so that load reduction targets and control actions can be reviewed if monitoring indicates continuing water quality problems.

PROBLEM IDENTIFICATION

A cover memo should describe the waterbody as it is identified on the state's section 303(d) list, the pollutant of concern, and the priority ranking of the waterbody. The TMDL submittal must include a description of the point, nonpoint, and natural background sources of the

TMDL Submittal Elements

Loading Capacity:	8.35 x 10^{12} counts/year (also with monthly allocations)
Load Allocation:	8.35 x 10^{12} counts/year (also with monthly allocations)
Wasteload Allocation:	8.34 x 10^{8} counts/day (0 percent reduction)
Seasonal Variation:	Monthly variation in source loading
Margin of Safety:	Implicit

[1] All information contained in this summary was obtained from MCTEW, 1999.

pollutant of concern, including the magnitude and location of the sources. The TMDL submittal should also contain a description of any important assumptions, such as (1) the assumed distribution of land uses in the watershed; (2) population characteristics, wildlife resources, and other relevant characteristics affecting pollutant characterization and allocation, as applicable; (3) present and future growth trends, if this factor was taken into consideration in preparing the TMDL; and (4) an explanation and analytical basis for expressing the TMDL through surrogate measures, if applicable.

Muddy Creek has been placed on Virginia's 303(d) list of water quality impaired waterbodies for fecal coliform bacteria, which are threatening the creek's designated uses. The state standard specifies that the maximum allowable level of fecal coliforms should not exceed 1,000 counts per 100 mL if only one sample is available for a 30-day period and a geometric mean allowable level should not exceed 200 counts per 100 mL if more than one sample is available for a 30-day period. A review of available monitoring data for the area indicates that fecal coliform bacteria are consistently above the 1,000 cfu/100 mL state standard. All waters of Virginia are designated for recreational uses; the propagation and growth of a balanced, indigenous population of aquatic life; wildlife; and the production of edible and marketable natural resources. The elevated levels of FC bacteria are threatening the use of Muddy Creek for recreational purposes.

DESCRIPTION OF THE APPLICABLE WATER QUALITY STANDARDS AND NUMERIC WATER QUALITY TARGET

The TMDL submittal must include a description of the applicable state water quality standard, including the designated use(s) of the waterbody, the applicable numeric or narrative water quality criterion, and the antidegradation policy. This information is necessary for EPA to review the load and wasteload allocation required by the regulation. A numeric water quality target for the TMDL (a quantitative value used to measure whether the applicable water quality standard is attained) must be identified. If the TMDL is based on a target other than a numeric water quality criterion, the submittal must include a description of the process used to derive the target.

For the Muddy Creek TMDL, the applicable endpoints and associated target values can be determined directly from the Virginia water quality standards. The in-stream fecal coliform target for this TMDL is an instantaneous maximum of 1,000 counts per 100 mL.

SOURCE ASSESSMENT

All potential sources of fecal coliform bacteria in the Muddy Creek watershed were identified based on an evaluation of current land use/cover, information on watershed activities (e.g., agricultural management activities), and discussions with local agency contacts. The source assessment was used as the basis of development of the model and ultimate analysis of the TMDL allocation options. The bacteria sources with the watershed included both point and nonpoint sources.

Two point sources were identified in EPA's Permit Compliance System (PCS) as discharging to Muddy Creek—Wampler Foods, Inc., and the Mount Clinton Elementary School. Wampler Foods is a poultry slaughtering and processing facility, and the school is a fairly small, intermittent, seasonal discharger. The Mount Clinton Elementary School was not included in the model analysis because it was scheduled for closure. Both of these sources discharge under a Virginia Pollutant Discharge Elimination System (VPDES) permit. PCS data were used to determine the maximum observed effluent concentrations and flow rates for Wampler Foods, which were used to represent the point source in the model.

To spatially analyze the bacteria loading, the Muddy Creek watershed was divided into eight subwatersheds. The land uses in each subwatershed were determined using National Aerial Photography Program (NAPP) and Farm Service Agency (FSA) aerial slides. The 24 land use classes in the watershed were grouped into 9 land use categories for the TMDL analysis in the Muddy Creek watershed. Each land use has various nonpoint sources that contribute fecal coliform bacteria to the land surface that potentially can be washed off into the receiving waters of the watershed. These nonpoint sources include failing septic systems and other uncontrolled discharges; wildlife; land application of liquid dairy manure; land application of poultry litter; cattle contributions directly deposited in-stream; and grazing animals. Extensive amounts of information on agricultural and management activities and watershed characteristics were obtained through coordination with state and regional agencies. This information was evaluated to identify, characterize, and quantify source contributions of fecal coliform bacteria. Information used to characterize the type, distribution, and behavior of sources in the Muddy Creek watershed included the following:

- Land use distributions (provided by Virginia Department of Conservation and Recreation [VADCR]).
- Livestock counts (provided by VADCR and confirmed by Soil and Water Conservation District [SWCD] and Natural Resources Conservation Service [NRCS] representatives).
- Information on cattle access to stream reaches (provided by VADCR and SWCD).
- Information on grazing and confinement schedules for cattle (provided by VADCR).
- Application rates and schedules for land application of liquid dairy manure (provided by VADCR).
- Literature values and site-specific information on characteristics of dairy manure (provided by VADCR).
- Application rates and schedules for land application of poultry litter (provided by VADCR and SWCD).
- Wildlife densities (provided by VADCR).
- Literature values on waste characteristics and fecal coliform bacteria production rates of various animals.
- Number of septic systems in the watershed and population served (provided by VADCR).
- Literature values for septic system failure rates for the county and discharge concentration and flow rate.

Based on their characteristics, nonpoint sources were represented in the analysis as either "direct" or "diffuse" sources. Runoff of accumulated fecal coliform from land uses was considered a diffuse source. Failing septic systems, straight pipes, and cattle contributing bacteria loads to stream reaches were considered direct sources discharging loads directly to stream reaches.

LOADING CAPACITY: LINKING WATER QUALITY AND POLLUTANT SOURCES

As described in EPA guidance, a TMDL describes the loading capacity of a waterbody for a particular pollutant. EPA regulations define loading capacity as the greatest amount of loading a waterbody can receive without violating water quality standards (40 CFR 130.2(f)). The TMDL submittal must describe the rationale for the analytical method used to establish the cause-and-effect relationship between the numeric target and the identified pollutant sources. In many circumstances, a *critical condition* must be described and related to physical conditions in the waterbody (40 CFR 130.7(c)(1)). Supporting documentation for the analysis must also be included, including the basis for assumptions, strengths and weaknesses in the analytical process, and results from water quality modeling, so that EPA can properly review the elements of the TMDL required by the statute and regulations.

The USEPA's Better Assessment Science Integrating Point and Nonpoint Sources (BASINS) system Version 2.0, with the Nonpoint Source Model (NPSM), was used to predict the significance of fecal coliform sources and fecal coliform levels in the Muddy Creek watershed. BASINS is a multipurpose environmental analysis system for use in performing watershed and water quality-based studies. A geographic information system (GIS) provides the integrating framework for BASINS and allows for the display and analysis of a wide variety of landscape information (e.g., land uses, monitoring stations, point source discharges). The NPSM model within BASINS simulates nonpoint source runoff from selected watersheds, as well as the transport and flow of the pollutants through stream reaches. Through calibration of model parameters and representation of watershed sources, the transport and delivery of bacteria to watershed streams and the resulting in-stream response and concentrations were simulated.

The hydrologic conditions in the Muddy Creek watershed are characterized by relatively random successions of dry, average, and wet rainfall years. A hydrologically representative time period used in modeling is necessary to account for the varying climatic and hydrologic conditions occurring within the watershed and to represent the potentially varying critical conditions. During dry weather and low flow, constant direct discharges dominate the impact on in-stream concentrations; however, during wet weather and high flow periods, surface runoff delivers nonpoint source fecal coliform to the stream, affecting the in-stream conditions more than constant discharges. To represent the varying meteorological conditions within the Muddy Creek watershed, analysts used a 5-year modeling period that covers a wide range of climatic and hydrologic conditions, allowing for a more accurate analysis of source loading and in-stream conditions within the Muddy Creek watershed.

Point and nonpoint sources were both represented in the model. Wampler Foods, Inc., was the only point source included in the model. Using the flow conditions provided by the Virginia Department of Environmental Quality (VADEQ), average flow and fecal coliform concentrations were calculated for Wampler. Fecal coliform accumulation rates (number/acre/day) were calculated for each land use based on all sources contributing fecal coliform to the land use.

The nonpoint sources identified in the watershed were represented in the model through build-up and wash-off processes or as "point" sources. For diffuse sources, fecal coliform accumulation rates (number/acre/day) were calculated for each land use based on all sources contributing fecal coliform bacteria to the land surface. For example, the fecal coliform accumulation rate for cropland is the sum of accumulation rates due to liquid dairy manure application, litter application, and deer. Accumulation rates for the agricultural land uses (Cropland, Pasture 1, Pasture 2, Pasture 3, and Loafing Lots) were calculated on a monthly basis to account for seasonal variations in litter and dairy manure application and grazing and confinement schedules for livestock. Literature values for typical fecal coliform production rates and the fecal coliform content of waste for various animals were used in the calculation of fecal coliform contributions from the various sources. Direct sources were represented in the modeling analysis as discharging directly to stream reaches with a characteristic flow and concentration for each month.

ALLOCATIONS

EPA regulations require that a TMDL include wasteload allocations (WLAs), which identify the portion of the loading capacity allocated to existing and future point sources (40 CFR 130.2(g)). If no point sources are present or the TMDL recommends a zero WLA for point sources, the WLA must be listed as zero. The TMDL may recommend a zero WLA if the state determines, after considering all pollutant sources, that allocating only to nonpoint sources will still result in attainment of the applicable water quality standard. In preparing the WLA, it is not necessary that every individual point source have a portion of the allocation of pollutant loading capacity. It is necessary, however, to allocate the loading capacity among individual point sources as necessary to meet the water quality standard. The TMDL submittal should also discuss whether a WLA is based on an assumption that loads from a nonpoint source or sources will be reduced. In such cases, the state needs to demonstrate reasonable assurance that the nonpoint source reductions will occur within a reasonable time.

EPA regulations also require that a TMDL include load allocations (LAs), which identify the portion of the loading capacity allocated to existing and future nonpoint sources and to natural background (40 CFR 130.2(h)). Load allocations may range from reasonably accurate estimates to gross allotments (40 CFR 130.2(g)). Where it is possible to separate natural background from nonpoint sources, separate LAs should be made and described. If there are neither nonpoint sources nor natural background or the TMDL recommends a zero LA, an explanation must be provided. The TMDL may recommend a zero LA if the state determines, after considering all pollutant sources, that allocating only to point sources will still result in attainment of the applicable water quality standard.

The statute and regulations require that a TMDL include a margin of safety to account for any lack of knowledge concerning the relationship between effluent limitations and water quality (CWA § 303(d)(1)(C), 40 CFR 130.7(c)(1)). EPA guidance explains that the MOS may be implicit, i.e., incorporated into the TMDL through conservative assumptions in the analysis, or explicit, i.e., expressed in the TMDL as loadings set aside for the MOS. If the MOS is implicit, the conservative assumptions in the analysis that account for the MOS must be described. If the MOS is explicit, the loading set aside for the MOS must be identified.

The statute and regulations require that a TMDL be established with seasonal variations. The state must describe the method chosen for including seasonal variations in the TMDL (CWA § 303(d)(1)(C), 40 CFR 130.7(c)(1)).

After conducting a sensitivity analysis on source impacts and developing allocation scenarios, a final TMDL was chosen. Because of the varying source characteristics and hydrologic conditions in the watershed, a combination of load reductions was necessary for both diffuse nonpoint sources (affecting water quality during high runoff/flow events) and direct nonpoint sources (affecting water quality during low flow and dilution events). Point sources in the watershed were considered negligible in their impact on in-stream fecal coliform levels, and their allocations were set equal to their existing load. Table B-1 presents the wasteload allocation for Wampler Foods, Inc.

Table B-2 presents the load allocations for nonpoint sources in the Muddy Creek watershed. These allocations represent the overall reductions from the fecal coliform sources for the year.

Land use activities and animal distribution vary between subwatersheds and from month to month within the Muddy Creek watershed. Monthly load allocations by subwatershed were presented as an appendix in the Muddy Creek TMDL report. Model simulation and representation of bacteria accumulation on a monthly basis and the resulting monthly load allocations account for seasonal variation in the TMDL analysis.

The margin of safety (MOS) is incorporated implicitly into the modeling process by setting the TMDL target 5 percent lower than the water quality standard of a geometric mean of 200 counts/100 mL. TMDL allocations were developed to meet a target of 190 counts/100 mL.

MONITORING PLAN

EPA's 1991 document *Guidance for Water Quality-Based Decisions: The TMDL Process* (EPA 440/4-91-001) calls for a monitoring plan when a TMDL is developed under the phased approach. The guidance provides that a TMDL developed under the phased approach also needs to provide assurances that nonpoint

Table B-1. Wasteload allocations to point sources in the Muddy Creek watershed

Point Source	Existing Load	Allocated Load	Percent Reduction
Wampler Foods, Inc.	8.34×10^8 counts/day	8.34×10^8 counts/day	0%

source control measures will achieve expected load reductions. The phased approach is appropriate when a TMDL involves both point and nonpoint sources and the point source WLA is based on an LA for which nonpoint source controls need to be implemented. Therefore, EPA's guidance provides that a TMDL developed under the phased approach should include a monitoring plan that describes the additional data to be collected to determine whether the load reductions required by the TMDL lead to attainment of water quality standards.

The state of Virginia will continue sampling for fecal coliform bacteria at two ambient monitoring stations to evaluate Muddy Creek's future compliance with water quality standards. Monthly sampling for fecal coliform bacteria will continue until the violation of the 1,000 counts/100 mL criterion is reduced to 10 percent or less. After this reduction, the monitoring frequency will increase to two or more samples within a 30-day period for evaluation of compliance with the 200 counts/100 mL geometric mean. The reason for this monitoring approach is that until the effects of the initial load reductions are reflected in lower fecal coliform counts in Muddy Creek, additional monthly samples will not provide additional information and the cost of the additional sampling is not justified. Two biological monitoring stations will also be sampled twice a year for benthic organisms.

IMPLEMENTATION PLANS

On August 8, 1997, EPA's Bob Perciasepe issued a memorandum, "New Policies for Establishing and Implementing Total Maximum Daily Loads (TMDLs)," which directs EPA regions to work in partnership with states to achieve nonpoint source load allocations established for 303(d)-listed waters impaired solely or primarily by nonpoint sources. To this end, the memorandum asks that the regions assist states in

Table B-2. Overall fecal coliform bacteria nonpoint source allocations for the Muddy Creek watershed for the representative hydrologic period

Source	Total Annual Loading for Existing Run (counts/year)	Total Annual Loading for Allocation Run (counts/year)	Percent Reduction
Diffuse nonpoint sources			
Built-up	1.88E+10	1.88E+10	0%
Farmstead	1.78E+10	1.78E+10	0%
Forest	7.33E+10	7.33E+10	0%
Barren	1.32E+08	1.32E+08	0%
Cropland	2.48E+11	2.16E+11	13.1%
Loafing lots	4.11E+12	8.08E+11	80.3%
Pasture 1	1.72E+12	1.01E+12	41.3%
Pasture 2	2.19E+11	1.28E+11	41.8%
Pasture 3	3.34E+12	1.94E+12	42.0%
Subtotal	9.75E+12	4.21E+12	56.8%
Direct nonpoint sources			
In-stream cattle	5.82E+14	4.14E+12	99.3%
Failing septic systems	7.72E+11	0	100%
Uncontrolled discharges	8.12E+13	0	100%
Subtotal	6.64E+14	4.14E+12	99.4%
TOTAL	6.73E+14	8.35E+12	98.8%

developing implementation plans that include reasonable assurances that the nonpoint source load allocations established in TMDLs for waters impaired solely or primarily by nonpoint sources will in fact be achieved; a public participation process; and recognition of other relevant watershed management processes. Although implementation plans are not approved by EPA, they help establish the basis for EPA's approval of TMDLs.

The state of Virginia will install a phased implementation process that allows for evaluation of the effectiveness of management practices and refinement of the model, as necessary. The target for the first phase of implementation in the Muddy Creek watershed will be a 10 percent or less violation of the 1,000 counts/100 mL instantaneous standard, achieved through the load allocations presented in Table B-3.

The VADEQ plans to incorporate TMDL implementation plans as part of the 303(e) Water Quality Management Plans (WQMPs). Virginia also administers many water quality-related programs, which will be used to support the Muddy Creek implementation plan. These programs include the Shenandoah-Potomac Tributary Strategy, the Watershed Restoration Action Strategy (WRAS) for the North River area, Virginia's Water Quality Improvement Fund, CWA and SDWA funding programs, and Virginia's agricultural cost share and incentives programs.

Table B-3. Overall Phase I fecal coliform bacteria nonpoint source allocations for the Muddy Creek watershed for the representative hydrologic period

Source	Total Annual Loading for Existing Run (counts/year)	Total Annual Loading for Allocation Run (counts/year)	Percent Reduction
Diffuse nonpoint sources			
Built-up	1.88E+10	1.88E+10	0%
Farmstead	1.78E+10	1.78E+10	0%
Forest	7.33E+10	7.33E+10	0%
Barren	1.32E+08	1.32E+08	0%
Cropland	2.48E+11	2.48E+11	0%
Loafing lots	4.11E+12	4.11E+12	0%
Pasture 1	1.72E+12	1.72E+12	0%
Pasture 2	2.19E+11	2.19E+11	0%
Pasture 3	3.34E+12	3.34E+12	0%
Subtotal	9.75E+12	9.75E+12	0%
Direct nonpoint sources			
In-stream cattle	5.8E+14	3.2E+13	94.4%
Failing septic systems	7.72E+11	0	100%
Uncontrolled discharges	8.12E+13	0	100%
Subtotal	6.64E+14	3.2E+13	94.4%
TOTAL	6.73E+14	4.18E+13	93.8%

REASONABLE ASSURANCES

EPA guidance calls for reasonable assurances when TMDLs are developed for waters impaired by both point and nonpoint sources or for waters impaired solely by nonpoint sources. In a water impaired by both point and nonpoint sources, where a point source is given a less stringent wasteload allocation based on an assumption that nonpoint source load reductions will occur, reasonable assurance must be provided for the TMDL to be approvable. This information is necessary for EPA to review the load allocations and wasteload allocations required by the regulation.

In a water impaired solely by nonpoint sources, reasonable assurances are not required for a TMDL to be approvable. For such nonpoint source-only waters, states are encouraged to provide reasonable assurances regarding achievement of load allocations in the implementation plans described in Section 7 of the protocol. As described in the August 8, 1997, Perciasepe memorandum, such reasonable assurances should be included in state implementation plans and "may be non-regulatory, regulatory, or incentive-based, consistent with applicable laws and programs."

Through the evaluation of a number of allocation scenarios, the Muddy Creek TMDL represents the most feasible TMDL for implementation. Load reductions from areas more difficult to control (e.g, cropland and pastureland) were minimized while reductions from areas where drainage and runoff control is more feasible (e.g., feedlots) were emphasized.

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KEY TO ACRONYMS

AGNPS	Agricultural Nonpoint Source Pollution Model
BASINS	Better Assessment Science Integrating Point and Nonpoint Sources
BMP	Best management practice
BOD	Biochemical Oxygen Demand
CE-QUAL-RIV1	Hydrodynamic and Water Quality Model for Streams
CE-QUAL-W2	Two-Dimensional, Laterally Averaged, Hydrodynamic and Water Quality Model
CFR	Code of Federal Regulations
CFU	Colony-forming units
CORMIX	Cornell Mixing Zone Expert System
CPP	Continuing planning process
CSO	Combined sewer overflow
CSS	Combined sewer system
CWA	Clean Water Act
CZARA	Coastal Zone Act Reauthorization Amendments
DO	Dissolved oxygen
DYNTOX	Dynamic Toxics Model
FC	Fecal coliform bacteria
FDA	Food and Drug Administration
FS	Fecal streptococci
GIS	Geographic Information System
HSPF	Hydrologic Simulation Program-Fortran
LA	Load allocation (for nonpoint sources in TMDLs)
MCLG	Maximum contaminant level goal
MOS	Margin of safety, a required TMDL element
MPN	Most probable number
NPDES	National Pollutant Discharge Elimination System
NPS	Nonpoint source
NRCS	Natural Resources Conservation Service
PCS	Permit Compliance System
POTW	Publicly-owned treatment works
PS	Point source
QUAL2E	The Enhanced Stream Water Quality Model
SDWA	Safe Drinking Water Act
SSO	Sanitary sewer overflow
SWMM	Storm Water Management Model
TC	Total coliform bacteria
TMDL	Total maximum daily load
USDA	United States Department of Agriculture
USEPA	United States Environmental Protection Agency
UV	Ultraviolet
WASP/TOXI5	Water Quality Analysis Simulation Program with a Toxic Submodel
WLA	Waste load allocation (for point sources in TMDLs)
WQS	Water quality standard
WWTP	Wastewater treatment plant

GLOSSARY

4Q3. A probability-based statistic representing the 4-day average low flow occurring once in 3 years.

7Q10. 7Q10 is the 7-day average low flow occurring once in 10 years; this probability-based statistic is used in determining stream design flow conditions and for evaluating the water quality impact of effluent discharge limits.

Activated Sludge. A biological solid (microorganisms) capable of stabilizing waste aerobically.

Advection. Bulk transport of the mass of discrete chemical or biological constituents by fluid flow within a receiving water. Advection describes the mass transport due to the velocity, or flow, of the waterbody.

Aerobic. Environmental conditions characterized by the presence of dissolved oxygen; used to describe biological or chemical processes that occur in the presence of oxygen.

Allocations. Allocations are that portion of a receiving water's loading capacity that is attributed to one of its existing or future sources (nonpoint or point) of pollution or to natural background sources. (Wasteload allocation (WLA) is that portion of the loading capacity allocated to an existing or future point source and a load allocation (LA) is that portion allocated to an existing or future nonpoint source or to natural background source. Load allocations are best estimates of the loading, which can range from reasonably accurate estimates to gross allotments, depending on the availability of data and appropriate techniques for predicting loading.)

Ambient water quality. Concentration of water quality constituent as measured within the waterbody.

Anaerobic. Environmental condition characterized by zero oxygen levels. Describes biological and chemical processes that occur in the absence of oxygen.

Anthropogenic. Pertains to the [environmental] influence of human activities.

Aquatic ecosystem. Complex of biotic and abiotic components of natural waters. The aquatic ecosystem is an ecological unit that includes the physical characteristics (such as flow or velocity and depth), the biological community of the water column and benthos, and the chemical characteristics such as dissolved solids, dissolved oxygen, and nutrients. Both living and nonliving components of the aquatic ecosystem interact and influence the properties and status of each component.

Assimilative capacity. The amount of pollutant load that can be discharged to a specific waterbody without exceeding water quality standards. Assimilative capacity is used to define the ability of a waterbody to naturally absorb and use a discharges substance without impairing water quality or harming aquatic life.

Bacteria. Single-celled microorganisms that lack a fully-defined nucleus and contain no chlorophyll. Bacteria of the coliform group are considered the primary indicators of fecal contamination and are often used to assess water quality.

BASINS (Better Assessment Science Integrating Point and Nonpoint Sources). A computer-run tool that contains an assessment and planning component that allows users to organize and display geographic information for selected watersheds. It also contains a modeling component to examine impacts of pollutant loadings from point and nonpoint sources and to characterize the overall condition of specific watersheds.

Benthic. Refers to material, especially sediment, at the bottom of an aquatic ecosystem. It can be used to describe the organisms that live on, or in, the bottom of a waterbody.

Best management practices (BMPs). Methods, measures, or practices that are determined to be reasonable and cost-effective means for a land owner to meet certain, generally nonpoint source, pollution control needs. BMPs include structural and nonstructural controls and operation and maintenance procedures.

Biochemical oxygen demand (BOD). The amount of oxygen per unit volume of water required to bacterially or chemically oxidize (stabilize) the oxidizable matter in water. Biochemical oxygen demand measurements are usually conducted over specific time intervals (5,10,20,30 days). The term BOD generally refers to a standard 5-day BOD test.

Calcareous. Pertaining to or containing calcium carbonate.

Calibration. The process of adjusting model parameters within physically defensible ranges until the resulting predictions give a best possible good fit to observed data.

Channel. A natural stream that conveys water; a ditch or channel excavated for the flow of water.

Clean Water Act (CWA). The Clean Water Act (formerly referred to as the Federal Water Pollution Control Act or Federal Water Pollution Control Act Amendments of 1972), Public Law 92-500, as amended by Public Law 96-483 and Public Law 97-117, 33 U.S.C. 1251 et seq. The Clean Water Act (CWA) contains a number of provisions to restore and maintain the quality of the nation's water resources. One of these provisions is section 303(d), which establishes the TMDL program.

Coastal Zone. Lands and waters adjacent to the coast that exert an influence on the uses of the sea and its ecology, or whose uses and ecology are affected by the sea.

Coliform bacteria. See Total coliform bacteria.

Combined sewer overflows (CSOs). Discharge of a mixture of stormwater and domestic waste when the flow capacity of a sewer system is exceeded during rainstorms. CSOs discharged to receiving water can result in contamination problems that may prevent the attainment of water quality standards.

Combined sewer system (CSS). Sewer system that receives both domestic wastewater and stormwater and conducts the mixture to a treatment facility.

Concentration. Amount of a substance or material in a given unit volume of solution. Usually measured in milligrams per liter (mg/l) or parts per million (ppm).

Contamination. Act of polluting or making impure; any indication of chemical, sediment, or biological impurities.

Cost-share program. Program that allocates project funds to pay a percentage of the cost of constructing or implementing a best management practice. The remainder of the costs are paid by the producer.

Critical condition. The combination of environmental factors that results in just meeting the water quality criterion and has an acceptably low frequency of occurrence.

Cross-sectional area. Wet area of a waterbody normal to the longitudinal component of the flow.

***Cryptosporidium*.** See protozoa.

Decay. Gradual decrease in the amount of a given substance in a given system due to various sink processes including chemical and biological transformation, dissipation to other environmental media, or deposition into storage areas.

Decomposition. Metabolic breakdown of organic materials; the by-products formation releases energy and simple organics and inorganic compounds. (See also Respiration.)

Designated uses. Those uses specified in water quality standards for each waterbody or segment whether or not they are being attained.

Deterministic model. A model that does not include built-in variability: same input will always equal the same output.

Die-off rate. The first-order decay rate for bacteria, pathogens, and viruses. Die-off depends on the particular type of water body (i.e. stream, estuary , lake) and associated factors that influence mortality.

Dilution. Addition of less concentrated liquid (water) that results in a decrease in the original concentration.

Direct runoff. Water that flows over the ground surface or through the ground directly into streams, rivers, and lakes.

Discharge. Flow of surface water in a stream or canal or the outflow of groundwater from a flowing artesian well, ditch, or spring. Can also apply to discharge of liquid effluent from a facility or to chemical emissions into the air through designated venting mechanisms.

Discharge permits (NPDES). A permit issued by the U.S. EPA or a state regulatory agency that sets specific limits on the type and amount of pollutants that a municipality or industry can discharge to a receiving water; it also includes a compliance schedule for achieving those limits. It is called the NPDES because the permit process was established under the National Pollutant Discharge Elimination System, under provisions of the Federal Clean Water Act.

Dispersion. The spreading of chemical or biological constituents, including pollutants, in various directions from a point source, at varying velocities depending on the differential instream flow characteristics.

Dissolved oxygen (DO). The amount of oxygen that is dissolved in water. It also refers to a measure of the amount of oxygen available for biochemical activity in a waterbody, and as an indicator of the quality of that water.

Dynamic model. A mathematical formulation describing the physical behavior of a system or a process and its temporal variability.

Ecosystem. An interactive system that includes the organisms of a natural community association together with their abiotic physical, chemical, and geochemical environment.

Effluent. Municipal sewage or industrial liquid waste (untreated, partially treated, or completely treated) that flows out of a treatment plant, septic system, pipe, etc.

Effluent limitation. Restrictions established by a state or EPA on quantities, rates, and concentrations in pollutant discharges.

Endpoint. An endpoint is a characteristic of an ecosystem that may be affected by exposure to a stressor. Assessment endpoints and measurement endpoints are two distinct types of endpoints that are commonly used by resource managers. An assessment endpoint is the formal expression of a valued environmental characteristic and should have societal relevance. A measurement endpoint is the expression of an observed or measured response to a stress or disturbance. It is a measurable environmental characteristic that is related to the valued environmental characteristic chosen as the assessment endpoint. The numeric criteria that are part of traditional water quality standards are good examples of measurement endpoints.

Enhancement. In the context of restoration ecology, any improvement of a structural or functional attribute.

Enteric. Of or within the gastrointestinal tract.

Enterococci. A subgroup of the fecal streptococci that includes *S. faecalis* and *S. faecium.* The enterococci are differentiated from other streptococci by their ability to grow in 6.5% sodium chloride, at pH 9.6, and at 10°C and 45°C. Enterococci are a valuable bacterial indicator for determining the extent of fecal contamination of recreational surface waters.

Epidemiology. All the elements contributing to the occurrence or non-occurrence of a disease in a population; ecology of a disease.

Escherichia coli. A subgroup of the fecal coliform bacteria. *E. coli* is part of the normal intestinal flora in humans and animals and is, therefore, a direct indicator of fecal contamination in a waterbody. The O157 strain, sometimes transmitted in contaminated waterbodies, can cause serious infection resulting in gastroenteritis. See Fecal coliform bacteria.

Estuarine number. Nondimensional parameter accounting for decay, tidal dispersion, and advection velocity. Used for classification of tidal rivers and estuarine systems.

Estuary. Brackish-water areas influenced by the tides where the mouth of the river meets the sea.

Existing use. Use actually attained in the waterbody on or after November 28, 1975, whether or not it is included in the water quality standards (40 CFR 131.3).

Fecal coliform bacteria. A subset of total coliform bacteria that are present in the intestines or feces of warm-blooded animals. They are often used as indicators of the sanitary quality of water. They are measured by running the standard total coliform test at an elevated temperature (44.5°C). Fecal coliform is approximately 20% of total coliform. See also Total coliform bacteria.

Fecal streptococci. These bacteria include several varieties of streptococci that originate in the gastrointestinal tract of warm-blooded animals such as humans (*Streptococcus faecalis*) and domesticated animals such as cattle (*Streptococcus bovis*) and horses (*Streptococcus equinus*).

Feedlot. A confined area for the controlled feeding of animals. Tends to concentrate large amounts of animal waste that cannot be absorbed by the soil and, hence, may be carried to nearby streams or lakes by rainfall runoff.

Flocculation. The process by which suspended colloidal or very fine particles are assembled into larger masses or floccules that eventually settle out of suspension.

Flux. Movement and transport of mass of any water quality constituent over a given period of time. Units of mass flux are mass per unit time.

Gastroenteritis. An inflammation of the stomach and the intestines.

Geochemical. Refers to chemical reactions related to earth materials such as soil, rocks, and water.

***Giardia lamblia*.** See protozoa.

Gradient. The rate of decrease (or increase) of one quantity with respect to another; for example, the rate of decrease of temperature with depth in a lake.

Groundwater. The supply of fresh water found beneath the earth's surface, usually in aquifers, which supply wells and springs. Because groundwater is a major source of drinking water, there is growing concern over contamination from leaching agricultural or industrial pollutants and leaking underground storage tanks.

Hot Spots. Locations in a waterbodies or sediments where hazardous substances have accumulated to levels which may pose risks to aquatic life, wildlife, fisheries, or human health.

Hydrology. The study of the distribution, properties, and effects of water on the earth's surface, in the soil and underlying rocks, and in the atmosphere.

Indicator. Measurable quantity that can be used to evaluate the relationship between pollutant sources and their impact on water quality.

Indicator organism. Organism used to indicate the potential presence of other (usually pathogenic) organisms. Indicator organisms are usually associated with the other organisms, but are usually more easily sampled and measured.

Infectivity. Ability to infect a host.

Initial mixing zone. Region immediately downstream of an outfall where effluent dilution processes occur. Because of the combined effects of the effluent buoyancy, ambient stratification, and current, the prediction of initial dilution can be involved.

Insolation. Exposure to the sun's rays.

Irrigation. Applying water or wastewater to land areas to supply the water and nutrient needs of plants.

Karst geology. Solution cavities and closely-spaced sinkholes formed as a result of dissolution of carbonate bedrock.

Land application. Discharge of wastewater onto the ground for treatment or reuse. (See: irrigation)

Leachate. Water that collects contaminants as it trickles through wastes, pesticides, or fertilizers. Leaching can occur in farming areas, feedlots, and landfills and can result in hazardous substances entering surface water, groundwater, or soil.

Load, Loading, Loading rate. The total amount of material (pollutants) entering the system from one or multiple sources; measured as a rate in weight per unit time.

Load allocation (LA). The portion of a receiving water's loading capacity that is attributed either to one of its existing or future nonpoint sources of pollution or to natural background sources. Load allocations are best estimates of the loading, which can range from reasonably accurate estimates to gross allotments, depending on the availability of data and appropriate techniques for predicting the loading. Wherever possible, natural and nonpoint source loads should be distinguished. (40 CFR 130.2(g))

Loading capacity (LC). The greatest amount of loading that a water can receive without violating water quality standards.

Low-flow. Stream flow during time periods where no precipitation is contributing to runoff to the stream and contributions from groundwater recharge are low. Low flow results in less water available for dilution of pollutants in the stream. Due to the limited flow, direct discharges to the stream dominate during low flow periods. Exceedences of water quality standards during low flow conditions are likely to be caused by direct discharges such as point sources, illicit discharges, and livestock or wildlife in the stream.

Margin of Safety (MOS). A required component of the TMDL that accounts for the uncertainty about the relationship between the pollutant loads and the quality of the receiving waterbody (CWA section 303(d)(1)(C)). The MOS is normally incorporated into the conservative assumptions used to develop TMDLs (generally within the calculations or models) and approved by EPA either individually or in state/EPA agreements. If the MOS needs to be larger than that which is allowed through the conservative assumptions, additional MOS can be added as a separate component of the TMDL (in this case, quantitatively, a TMDL = LC = WLA + LA + MOS).

Mass balance. An equation that accounts for the flux of mass going into a defined area and the flux of mass leaving the defined area. The flux in must equal the flux out.

Mass loading. The quantity of a pollutant transported to a waterbody.

Mathematical model. A system of mathematical expressions that describe the spatial and temporal distribution of water quality constituents resulting from fluid transport and the one, or more, individual processes and interactions within some prototype aquatic ecosystem. A mathematical water quality model is used as the basis for waste load allocation evaluations.

Meningitis. Inflammation of the meninges, especially as a result of infection by bacteria or viruses.

Mitigation. Actions taken to avoid, reduce, or compensate for the effects of environmental damage. Among the broad spectrum of possible actions are those which restore, enhance, create, or replace damaged ecosystems.

Monitoring. Periodic or continuous surveillance or testing to determine the level of compliance with statutory requirements and/or pollutant levels in various media or in humans, plants, and animals.

Monte Carlo simulation. A stochastic modeling technique that involves the random selection of sets of input data for use in repetitive model runs. Probability distributions of receiving water quality concentrations are generated as the output of a Monte Carlo simulation.

National Pollutant Discharge Elimination System (NPDES). The national program for issuing, modifying, revoking and reissuing, terminating, monitoring, and enforcing permits, and imposing and enforcing pretreatment requirements, under Sections 307, 402, 318, and 405 of the Clean Water Act.

Natural background levels. Natural background levels represent the chemical, physical, and biological conditions that would result from natural geomorphological processes such as weathering or dissolution.

Natural waters. Flowing water within a physical system that has developed without human intervention, in which natural processes continue to take place.

Nonpoint source. Pollution that is not released through pipes but rather originates from multiple sources over a relatively large area. Nonpoint sources can be divided into source activities related to either land or water use including failing septic tanks, improper animal-keeping practices, forest practices, and urban and rural runoff.

Numeric Targets. A measurable value determined for the pollutant of concern which is expected to result in the attainment of water quality standards in the listed waterbody.

Organic matter. The organic fraction that includes plant and animal residue at various stages of decomposition, cells and tissues of soil organisms, and substance synthesized by the soil population. Commonly determined as the amount of organic material contained in a soil or water sample.

Outfall. Point where water flows from a conduit, stream, or drain.

Oxidation. The chemical union of oxygen with metals or organic compounds accompanied by a removal of hydrogen or another atom. It is an important factor for soil formation and permits the release of energy from cellular fuels.

Oxidation pond. A relatively shallow body of wastewater contained in an earthen basin; lagoon; stabilization pond.

Oxygen demand. Measure of the dissolved oxygen used by a system (microorganisms) in the oxidation of organic matter. See also biochemical oxygen demand.

Partition coefficients. Chemicals in solution are partitioned into dissolved and particulate adsorbed phase based on their corresponding sediment-to-water partitioning coefficient.

Pathogen. Disease-causing agent, especially microorganisms such as bacteria, protozoa, and viruses.

Permit. An authorization, license, or equivalent control document issued by EPA or an approved federal, state, or local agency to implement the requirements of an environmental regulation; e.g., a permit to operate a wastewater treatment plant or to operate a facility that may generate harmful emissions.

Permit Compliance System (PCS). Computerized management information system which contains data on NPDES permit-holding facilities. PCS keeps extensive records on more than 65,000 active water-discharge permits on sites located throughout the nation. PCS tracks permit, compliance, and enforcement status of NPDES facilities.

Phased approach. Under the phased approach to TMDL development, LAs and WLAs are calculated using the best available data and information recognizing the need for additional monitoring data to accurately characterize sources and loadings. The phased approach is typically employed when nonpoint sources dominate. It provides for the implementation of load reduction strategies while collecting additional data.

Point source. Pollutant loads discharged at a specific location from pipes, outfalls, and conveyance channels from either municipal wastewater treatment plants or industrial waste treatment facilities. Point sources can also include pollutant loads contributed by tributaries to the main receiving water stream or river.

Pollutant. Dredged spoil, solid waste, incinerator residue, sewage, garbage, sewage sludge, munitions, chemical wastes, biological materials, radioactive materials, heat, wrecked or discarded equipment, rock, sand, cellar dirt and industrial, municipal, and agricultural waste discharged into water. (CWA Section 502(6)).

Pollution. Generally, the presence of matter or energy whose nature, location, or quantity produces undesired environmental effects. Under the Clean Water Act, for example, the term is defined as the man-made or man-induced alteration of the physical, biological, chemical, and radiological integrity of water.

Pretreatment. The treatment of wastewater to remove or reduce contaminants prior to discharge into another treatment system or a receiving water.

Primary treatment. A basic wastewater treatment method that uses settling, skimming, and (usually) chlorination to remove solids, floating materials, and pathogens from wastewater. Primary treatment typically removes about 35 percent of biochemical oxygen demand (BOD) and less than half of the metals and toxic organic substances.

Protozoa. Single-celled organisms that reproduce by fission and occur primarily in the aquatic environment. Waterborne pathogenic protozoans of primary concern include *Giardia lamblia* and *Cryptosporidium*, both of which affect the gastrointestinal tract.

Public comment period. The time allowed for the public to express its views and concerns regarding action by EPA or states (e.g., a *Federal Register* notice of a proposed rule-making, a public notice of a draft permit, or a Notice of Intent to Deny).

Publicly Owned Treatment Works (POTW). Any device or system used in the treatment (including recycling and reclamation) of municipal sewage or industrial wastes of a liquid nature that is owned by a state or municipality. This definition includes sewers, pipes, or other conveyances only if they convey wastewater to a POTW providing treatment.

Raw sewage. Untreated municipal sewage.

Receiving waters. Creeks, streams, rivers, lakes, estuaries, groundwater formations, or other bodies of water into which surface water and/or treated or untreated waste are discharged, either naturally or in man-made systems.

Residence time. Length of time that a pollutant remains within a section of a waterbody. The residence time is determined by the streamflow and the volume of the river reach or the average stream velocity and the length of the river reach.

Respiration. Biochemical process by means of which cellular fuels are oxidized with the aid of oxygen to permit the release of the energy required to sustain life; during respiration, oxygen is consumed and carbon dioxide is released.

Restoration. Return of an ecosystem to a close approximation of its condition prior to disturbance.

Riparian zone. The border or banks of a stream. Although this term is sometimes used interchangeably with floodplain, the riparian zone is generally regarded as relatively narrow compared to a floodplain. The duration of flooding is generally much shorter, and the timing less predictable, in a riparian zone than in a river floodplain.

Runoff. That part of precipitation, snow melt, or irrigation water that runs off the land into streams or other surface water. It can carry pollutants from the air and land into receiving waters.

Safe Drinking Water Act. The Safe Drinking Water Act authorizes EPA to set national health-based standards for drinking water to protect against both naturally occurring and man-made contaminants that may be found in drinking water. EPA, states, and water systems then work together to make sure these standards are met.

Sanitary sewer overflow (SSO). When wastewater treatment systems overflow due to unforseen pipe blockages or breaks, unforseen structural, mechanical, or electrical failures, unusually wet weather conditions, insufficient system capacity, or a deteriorating system.

Scoping modeling. Involves simple, steady-state analytical solutions for a rough analysis of the problem.

Scour. To abrade and wear away. Used to describe the weathering away of a terrace or diversion channel or streambed. The clearing and digging action of flowing water, especially the downward erosion by stream water in sweeping away mud and silt on the outside of a meander or during flood events.

Secondary treatment. The second step in most publicly owned waste treatment systems, in which bacteria consume the organic parts of the waste. It is accomplished by bringing together waste, bacteria, and oxygen in trickling filters or in the activated sludge process. This treatment removes floating and settleable solids and about 90 percent of the oxygen-demanding substances and suspended solids. Disinfection is the final stage of secondary treatment. (See primary, tertiary treatment.)

Sediment. Organic or inorganic material often suspended in liquid that eventually settles to the bottom.

Sedimentation. Deposition or settlement of suspended matter in water, wastewater, or other liquids.

Septic system. An on-site system designed to treat and dispose of domestic sewage. A typical septic system consists of a tank that receives waste from a residence or business and a system of tile lines or a pit for disposal of the liquid effluent (sludge) that remains after decomposition of the solids by bacteria in the tank; must be pumped out periodically.

Sewer. A channel or conduit that carries wastewater and stormwater runoff from the source to a treatment plant or receiving stream. "Sanitary" sewers carry household, industrial, and commercial waste. "Storm" sewers carry runoff from rain or snow. "Combined" sewers handle both.

Simulation. Refers to the use of mathematical models to approximate the observed behavior of a natural water system in response to a specific known set of input and forcing conditions. Models that have been validated, or verified, are then used to predict the response of a natural water system to changes in the input or forcing conditions.

Slope. The degree of inclination to the horizontal. Usually expressed as a ratio, such as 1:25 or 1 on 25, indicating one unit vertical rise in 25 units of horizontal distance, or in a decimal fraction (0.04); degrees (2 degrees 18 minutes), or percent (4 percent).

Sorption. The adherence of ions or molecules in a gas or liquid to the surface of a solid particle with which they are in contact.

Stakeholder. Those parties likely to be affected by the TMDL.

Steady-state model. Mathematical model of fate and transport that uses constant values of input variables to predict constant values of receiving water quality concentrations.

STORET. U.S. Environmental Protection Agency (EPA) national water quality database for STORage and RETrieval (STORET). Mainframe water quality database that includes physical, chemical, and biological data measured in waterbodies throughout the United States.

Storm runoff. Stormwater runoff, snowmelt runoff, and surface runoff and drainage; rainfall that does not evaporate or infiltrate the ground because of impervious land surfaces or a soil infiltration rate lower than rainfall intensity, but instead flows onto adjacent land or waterbodies or is routed into a drain or sewer system.

Stormwater. The portion of precipitation that does not naturally percolate into the ground or evaporate, but flows via overland flow, interflow, channels or pipes into a defined surface water channel, or a constructed infiltration facility.

Stormwater management models (SWMM). USEPA mathematical model that simulates the hydraulic operation of the combined sewer system and storm drainage sewershed.

Stratification (of waterbody). Formation of water layers each with specific physical, chemical, and biological characteristics. As the density of water decreases due to surface heating, a stable situation develops with lighter water overlaying heavier and denser water.

Stressor. Any physical, chemical, or biological entity that can induce an adverse response.

Surface runoff. Precipitation, snowmelt, or irrigation water in excess of what can infiltrate the soil surface and be stored in small surface depressions; a major transporter of nonpoint source pollutants.

Surface water. All water naturally open to the atmosphere (rivers, lakes, reservoirs, ponds, streams, impoundments, seas, estuaries, etc.) and all springs, wells, or other groundwater collectors directly influenced by surface water.

Suspended solids or load. Organic and inorganic particles (sediment) suspended in and carried by a fluid (water). The suspension is governed by the upward components of turbulence, currents, or colloidal suspension. Suspended sediment usually consists of particles <0.1 mm, although size may vary according to current hydrological conditions. Particles between 0.1 mm and 1 mm may move as suspended or bedload.

Technology-based limits. Industry-specified effluent limitations applied to a discharge when it will not cause a violation of water quality standards at low stream flows. Usually applied to discharges into large rivers.

Tertiary treatment. Advanced cleaning of wastewater that goes beyond the secondary or biological stage, removing nutrients such as phosphorus, nitrogen, and most biochemical oxygen demand (BOD) and suspended solids.

Three-dimensional model (3-D). Mathematical model defined along three spatial coordinates where the water quality constituents are considered to vary over all three spatial coordinates of length, width, and depth.

Topography. The physical features of a surface area including relative elevations and the position of natural and man-made features.

Total coliform bacteria. A particular group of bacteria, found in the feces of warm-blooded animals, that are used as indicators of possible sewage pollution. They are characterized as aerobic or facultative anaerobic, gram-negative, nonspore-forming, rod-shaded bacteria which ferment lactose with gas formation within 48 hours at 35°. Note that many common soil bacteria are also total coliforms, but do not indicate fecal contamination. See also fecal coliform bacteria.

Total Maximum Daily Load (TMDL). The sum of the individual wasteload allocations (WLAs) for point sources, load allocations (LAs) for nonpoint sources and natural background, and a margin of safety (MOS). TMDLs can be expressed in terms of mass per time, toxicity, or other appropriate measures that relate to a state's water quality standard.

Toxic substances. Those chemical substances which can potentially cause adverse affects on living organisms. Toxic substances include pesticides, plastics, heavy metals, detergent, solvent, or any other materials that are poisonous, carcinogenic, or otherwise directly harmful to human health and the environment as a result of dose or exposure concentration and exposure time. The toxicity of toxic substances is modified by variables such as temperature, chemical form, and availability.

Tributary. A lower order stream compared to a receiving waterbody. "Tributary to" indicates the largest stream into which the reported stream or tributary flows.

Turbidity. The amount of light that is scattered or absorbed by a fluid.

Two-dimensional model (2-D). Mathematical model defined along two spatial coordinates where the water quality constituents are considered averaged over the third remaining spatial coordinate. Examples of 2-D models include descriptions of the variability of water quality properties along: (a) the length and width of a river that incorporates vertical averaging or (b) length and depth of a river that incorporates lateral averaging across the width of the waterbody.

Unstratified. Indicates a vertically uniform or well-mixed condition in a waterbody. See also Stratification.

Urban runoff. Water containing pollutants like oil and grease from leaking cars and trucks; heavy metals from vehicle exhaust; soaps and grease removers; pesticides from gardens; domestic animal waste; and street debris, which washes into storm drains and enters surface waters.

Validation (of a model). Process of determining how well the mathematical representation of the physical processes of the model code describes the actual system behavior.

Verification (of a model). Testing the accuracy and predictive capabilities of the calibrated model on a data set independent of the data set used for calibration.

Virus. Submicroscopic pathogen consisting of a nucleic acid core surrounded by a protein coat. Requires a host in which to replicate (reproduce).

Wasteload allocation (WLA). The portion of a receiving water's loading capacity that is allocated to one of its existing or future point sources of pollution. WLAs constitute a type of water quality-based effluent limitation (40 CFR 130.2(h)).

Wastewater. Usually refers to effluent from a sewage treatment plant.

Wastewater treatment. Chemical, biological, and mechanical procedures applied to an industrial or municipal discharge or to any other sources of contaminated water in order to remove, reduce, or neutralize contaminants.

Water quality. The biological, chemical, and physical conditions of a waterbody. It is a measure of a waterbody's ability to support beneficial uses.

Water quality criteria. Elements of state water quality standards expressed as constituent concentrations, levels, or narrative statement, representing a quality of water that supports a particular use. When criteria are met, water quality will generally protect the designated use.

Water quality standard. State or federal law or regulation consisting of a designated use or uses for the waters of the United States, water quality criteria for such waters based upon such uses, and an antidegradation policy and implementation procedures. Water quality standards protect the public health or welfare, enhance the quality of water and serve the purposes of the Clean Water Act.

Watershed. A drainage area or basin in which all land and water areas drain or flow toward a central collector such as a stream, river, or lake at a lower elevation.

Wetlands. An area that is constantly or seasonally saturated by surface water or groundwater with vegetation adapted for life under those soil conditions, as in swamps, bogs, fens, marshes, and estuaries.